Beyond the design event: Sediment pollution movement in SuDS

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Abstract

Sustainable urban Drainage Systems (SuDS) present a ‘blue-green’ method of urban stormwater management that is increasingly implemented in the UK and worldwide. SuDS mimic natural vegetated flow paths and are designed to manage the increase in stormwater quantity and degradation of stormwater quality resulting from development (urbanisation). They have been widely implemented across the UK over the last 15 years, to aid compliance with the EU Water Framework Directive (2000) standards for river water quality. Given the increasing maturity of UK SuDS, there is growing concern over the long-term performance efficiencies of these assets/networks, particularly the variability of treatment efficiency over multiple flow events. Providing the field monitoring evidence base to address this concern forms the aim of the present thesis. Emphasis is placed upon understanding SuDS asset/network sediment detention efficiency, as the majority of urban pollution is adsorbed to sediment material rather than transported and treated as solute. A novel tracer method is, therefore, developed and employed to identify and quantify sediment processes for mature UK case study SuDS.

SuDS design manuals (CIRIA 2015, Water by Design 2006, USEPA 2009) present expected or reported sediment detention and pollution mitigation levels for specific SuDS assets. For example, the expected Total Suspended Solids (TSS) removal for a swale has been defined as 90% in the WSUD (Water Sensitive Urban Design) technical manual (Leisenring et al. 2013). Yet, these treatment efficiencies are based on a single ‘design’ rainfall-runoff event through the system; hence, fail to consider the sensitivity of SuDS performance to non-design and multiple repeat events over the long term design life of the SuDS asset. As natural variability in rainfall affects pollutant washoff, shear stress for entrainment, conveyance, deposition, loss of treatment capacity etc., the research presented in the present thesis intensively monitored four established UK SuDS networks for 6-12 month timeframes. Bulk sampling data highlight that TSS treatment is highly variable, ranging from highly effective (>80%) to inefficient (<20%). Similar variability is found in sediment deposition rates (on average: 0.4-1.1 kg/m^2/yr), providing insight on temporal and asset dependency of fine sediment detention, including related treatment efficiency and long term loss of capacity. Wetlands illustrate the most effective (mass) sediment detention per area (>1kg/m^2) while the swales detain the least (<0.8kg/m^2).

To advance the volumetric data noted above, source-sink routing of diffuse fine sediment pollution required development of tracer methodology appropriate to use in SuDS. This dictated use of Rare Earth Oxides (REO) as fine sediment tags; although their use in an urban environment is new, it provides long term trace and experimental replicability results without loss of provenance, signature degradation or loss of tag material. Thus, unique time-stamped and source-specific identifiers have been used monitor their movement into and through each SuDS over a 6-12 month period. Use of REO tagged sediment data permits mass balance analysis of fine sediment through the monitored SuDS assets and networks. Data clearly illustrate that sediment is not fully detained (as assumed in SuDS design); rather, sediment is re-entrained and re-deposited multiple times over multiple flow events. Residence time of sediment within a full SuDS network is found to be as short as 12 weeks, raising concern over treatment capability. Reviewing this at finer asset-based resolution, detention efficiency and conveyance rates appear unique to each asset. Generally linear wetland and swale assets demonstrate the greatest (tagged) sediment detention efficiency (>70%) while (the monitored) wetland assets decline to below 50% efficiency over the first 12 months and ponds demonstrated negligible sediment detention efficiency (<10%).

As 80% of urban pollution is conveyed adsorbed to fine sediment, the sediment conveyance pattern through SuDS assets has been analysed to define the pollutant concentration levels and trends of detained sediment. Pollutant levels show no consistent trend across SuDS assets. Results illustrate that sediment pollutant contamination shows an influence from particle size distribution and mass deposition as well as asset design. Analysis shows the most numerous significantly elevated sediment pollutant concentrations within the linear wetland, with Fe, Ba, Cr, Cu, Zn, K and P demonstrating average concentrations above contaminated land thresholds. Enrichment and geoaccumulation indexing of pollutants illustrates Fe, Zn, Cr, Ba, Cu and P to be pollutants of concern, with Fe, Pb, Ni, Cr, Mn, Zn, Ba, Cu, Ni and P identified as hotspot pollutants in one or multiple SuDS assets.

Cross-correlation of rainfall and flow characteristics with asset/network detention efficiencies were used to define key drivers of multi-event sediment conveyance. Outcomes highlight three variables of strong influence: the number of rainfall and flow occurrences; the antecedent dry days; rainfall clustering characteristics. Weaker correlations are found with flow characteristics (number of flow events, depth and velocity – leading to Fr and R_e values) and modal particle size. These influential forcings have then been considered with respect to a selected standard SuDS pollution treatment process (MUSIC k-C*) to identify the compatibility for multiple rainfall-runoff event SuDS fine sediment and pollutant simulation. The research provides multiple event SuDS stormwater treatment efficiencies that can inform improved SuDS design and maintenance planning by engineering consultants, Local Authorities, environmental regulators and SuDS asset managers.

Dedication

This thesis is dedicated to Steve Allen, my amazing husband, for the love, patience, time, inspiration and support he gave me. You helped me survive what I thought was impossible. This is as much yours as it could ever be mine. I could not have done this without you.

ACADEMIC REGISTRY
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1 Research Introduction

1.1 Introduction to current SuDS understanding and the research focus

Sustainable urban Drainage Systems (SuDS) present a ‘blue-green’ method of urban stormwater management that is increasingly implemented in the UK and worldwide (Lawson et al 2014). SuDS mimic natural vegetated flow paths and are designed to manage the increase in stormwater quantity and degradation of stormwater quality resulting from development (urbanisation). The EU Water Framework Directive (2000) has provided a benchmark and standards for river water quality. SuDS have been increasingly incorporated into urban developments within Scotland and the wider UK over the past 20 years to help meet these standards. However, to quantify the benefits of SuDS in achieving healthy rivers and in mitigating the impact of urbanisation on stormwater quality it is necessary to understand the treatment capacity, mitigation and management measures provided by SuDS.

Urbanisation elevates the hazard of sediment pollution by both, an increase in the washoff quantity of sediment <2mm and, a rise in metal/mineral sediment concentrations as a result of urban activities. Up to 85% of urban pollution is adsorbed to fine sediment, illustrating this material to be a key carrier and factor in urban stormwater pollution management. Previous SuDS field and laboratory studies have identified the event based suspended solid and solute pollutant removal efficiencies (Davis et al. 2001, Heal et al. 2006, McNett and Hunt 2011). Thus there is an understanding of the solute pollution change or water quality improvement provided by SuDS, and this is reflected in current design guidance (CIRIA 2015, Water by Design 2006). However, few studies have been completed to define the conveyance, deposition and detention of fine sediment (and the associated pollutants) within SuDS assets or networks.

To define the capacities for water quality improvement and the efficiencies of the SuDS assets/networks it is important to consider the pollutant conveyance process for the defined life cycle of these assets. SuDS are designed to function for 20-30 years if provided with adequate maintenance (after which restoration or remediation activities may be required). Thus, the influence of multiple consecutive rainfall-runoff events over an extended time period must be defined to accurately examine long term movement, detention efficiency and water quality remediation of sediment through SuDS assets. Examining the research literature identifies that these are current knowledge gaps (i.e. the quantification of sediment detention within SuDS, and the pollutant levels of this material) forming a critical oversight in SuDS design manuals (e.g. CIRIA 2015, Water by Design 2006, USEPA 2009). These manuals have been developed explicitly to inform SuDS asset design and to define expected sediment detention and pollution mitigation levels for each specific type of SuDS assets. For example, the expected total suspended solids removal for a swale has been defined as 90% in the WSUD technical manual (Leisenring et al. 2013). Yet, the calculated sediment and pollution stormwater treatment efficiencies are based on single design rainfall-runoff events. The influence of multiple events over the effective life cycle of the asset is not examined, evidenced (e.g. monitoring) or included in the definition of current design standard 'ideal' treatment efficiencies.

The need for this long term sediment detention and water quality remediation information is driven by EU Directive requirements and by Local Authorities concern over SuDS maintenance planning and operation. Where high sediment detention is expected, there is a resultant corresponding loss of stormwater storage volume. The loss in storage volume increases flood risk and reduces resilience to infrastructure downstream of the network. To minimise this problem, it is important that the Local Authorities maintain the designed water capacity (flood storage) of SuDS assets. This requires methodology towards estimating their sediment detention efficiencies by which to calculate the volume and frequency of remediation by excavation practices.

An added complexity is that the sediment deposited within SuDS is likely to be polluted. The majority of urban pollutants are adsorbed to sediment (Jones et al. 2008) and are, potentially, deposited within SuDS assets. Bioremediation, the use of plants to extract and/or remove pollutants within stormwater and sediment, is a key component to maintain the water treatment objective of SuDS design, thus increasing water quality of SuDS effluent. The literature shows notable recent research advancing the understanding of phytoremediation processes, particularly focused on infiltration and wetting/drying cycles on ephemeral vegetated surfaces. Yet, the understanding of sediment provision, residency and transport has not been incorporated into these studies, despite directly influencing the pollutant loading and residence time for remediation. This incomplete scientific knowledge has resulted in hesitancy by Local Authorities in sediment removal in SuDS maintenance activities, due to the potential toxicity of this material requiring safe disposal or further treatment (incurring added cost) post-removal.

To address these deficiencies, the aim of the presented research is to provide field-based evidence from a number of SuDS networks to quantify the long term sediment detention efficiencies of SuDS assets and provide new insight into sediment pollution, appropriate to better informed design and maintenance strategy.

1.2 Research objectives

To meet the aim stated in Section 1.1, the research objectives of this thesis focus on field-based monitoring and analysis of sediment movement (TSS and deposition) and pollutant concentrations within established SuDS over extended (multiple rainfall-runoff event) periods of 6-12months. Within this research, all sediment ≤ 2mm is defined as 'fine' urban sediment. However, it is acknowledged that this definition includes clay (the traditional 'fine' sediment classification, ≤ 63µm), silt and find sand sized material. The term 'fine sediment' has been used to simply indicate the sediment of particle size within the research focus. Specifically:

(1) To quantify the long term sediment detention efficiencies of SuDS assets and networks:

- To identify if fine sediment is retained or detained within SuDS assets;
- To ascertain if deposited sediment becomes resuspended by subsequent rainfall-runoff events;
- To quantify the detention efficiency of selected SuDS assets;
- To advance scientific knowledge of long term SuDS sediment detention (quantification) and provide evidenced based projections of life-cycle (25-30 year) sediment detention.

(2) To identify key influences and parameters of SuDS sediment pollution detention efficiency:

- To determine which meteorological (rainfall), hydraulic (flow) and sediment characteristics or parameters influence SuDS detention efficiency;
- To quantify the temporal influence on SuDS detention efficiency;
- To review and/or develop statistical descriptions of SuDS sediment detention efficiency that are able to describe multiple event influence.

(3) To examine the contamination level of deposited sediment within SuDS assets:

- To identify any temporal (seasonal) trends in pollutant concentration and remediation;

- To provide a SuDS pollutant detention and remediation efficiency field dataset for urban-sourced elements beyond those commonly tested in water quality assessment i.e. heavy metals (Pb, Zn, Ni, Cu) and nutrients (P);
- To compare SuDS asset performance in the remediation of individual pollutants.

(4) To achieve research Objectives 1-3 it is also necessary to develop a sediment tracer methodology that can function conservatively in the SuDS environment, over an extended monitoring period and which allows for adequate replication of field based experiments.

By achieving these objectives, the outputs of this thesis seek to provide a detailed critical review of SuDS design guidance relating to sediment detention efficiency, intended to improve future design, guide future maintenance needs and provide new knowledge (and related advice) on the potential occurrence and contamination concentration of detained material within SuDS.

1.3 Thesis structure

The thesis has been constructed to present a research story, structured into the following Chapters:

Chapter 1: Research Introduction

Chapter 1 presents a short summary of current context, knowledge and research needs regarding SuDS sediment detention and pollutant remediation. This introduces the overall aim and key research objectives underpinning the research presented herein.

Chapter 2: Literature Review

The literature review has been constructed to achieve two objectives. Firstly, to present necessary background on the current state of research knowledge (and its real-world application) of urban pollution, SuDS efficiencies, sediment transport and pollution remediation. Secondly, the literature review provides detailed identification of the research gaps in current knowledge and the need for further research.

Chapter 3: Evolution of a novel tracer methodology

Chapter 3 presents the methodology behind the research undertaken and presented within this thesis. The field sites and SuDS assets examined and monitored within this research are presented alongside the field sampling and laboratory analysis. The novel sediment tracer method designed and used to monitor the movement of fine sediment through the SuDS assets and networks is described here in detail, including initial field-based pilot tests and iterative methodological revisions during development of this technique.

Chapter 4: Mass sediment movement and deposition within the SuDS assets and networks

The source - pathway – receptor process of sediment movement affecting SuDS is presented in Chapter 4. For the case studies monitored, urban sediment supply is compared to literature reported results for different urban land-uses. The focus of Chapter 4 is the quantification of mass sediment deposition within the monitored SuDS assets over the extended monitoring period, and the simultaneous total suspended sediment treatment efficiencies for the SuDS assets. The deposition and suspended sediment efficiencies are examined (comparison of influent to effluent) to identify the occurrence of high (>80%), functional (20-80%) or inefficient (<20%) detention or suspended sediment reduction. Correlation analysis is presented to identify influential rainfall, flow and sediment characteristics on total sediment detention efficiencies.

Chapter 5: Sediment transport through SuDS assets over multiple events

Chapter 5 presents the fine sediment tracer results for sediment deposition and conveyance through the monitored SuDS. Using the novel REO tracer methodology a simple mass balance analysis of fine sediment is undertaken. The multiple event sediment detention efficiency of each SuDS asset and the network is also presented.

Chapter 6: SuDS asset sediment drivers empirical analysis

The rainfall, flow and sediment characteristics are cross-correlated with fine sediment detention to establish those with strong correlation results (see also Appendix VI). This analysis highlights influential drivers of SuDS sediment detention efficiency, which are then used to provide a statistical description (regression function) of influential characteristics of SuDS sediment detention efficiency.

Chapter 7: Sediment contamination within SuDS

Chapter 7 examines the urban pollutant concentration in deposited sediment via ICP-OES analysis. The levels of contamination within the SuDS assets (sampled quarterly) are compared to literature-based evidence. A wider range of urban pollutants are considered within this Chapter than previously published; trace elements and urban pollutants (organic matter analysis was beyond the scope of this study due to experimental constraints). Urban pollutant concentrations (adsorbed to sediment) are considered alongside total sediment deposition and movement, and a new method of identifying sediment pollutant contamination of concern, considering both actual and enrichment levels, is presented.

Chapter 8: Conclusions, limitations and future research

Chapter 8 provides a summary of the key research findings from this thesis. The novelty and impact of this research is discussed, within the focus of SuDS design, guidance and maintenance improvement but also within the greater view of water quality and sediment transport research. The research limitations are discussed within this Chapter and future research opportunities, identified through this thesis research and findings, are presented.

2 Literature Review

2.1 Introduction

Sustainable urban drainage has evolved from the need to manage urban runoff for water quality and quantity purposes (Butler and Parkinson 1997, Fletcher et al. 2015). Historically, runoff from urban development was collected via surface inlets and conveyed (often by subsurface combined sewer pipe networks) to downstream receiving waterways as part of the cities combined sewer outflow (stormwater mixed with foul water) (Delleur 2003, Tibbets 2005). As cities have expanded and intensified, the quantity of runoff has increased, resulting in larger infrastructure to collect, convey and treat this stormwater+foul urban waste water. In response to the increasing demand on infrastructure, risk of overflow during high rainfall events and in acknowledgement that urban stormwater runoff contains different pollutants from foul waste, separate sewer systems became best practice in new urban development (Tibbets 2005, Butler and Davis 2009, Harremös 2002). Stormwater drains were thus designed to collect and convey urban runoff directly to the receiving waterways, without any flow control or water quality treatment. More recently, since the acknowledgement of urban diffuse pollution impact on receiving waterway and ecosystems, and with the advent of legislation to control the pollution levels in stormwater discharge (Urban Water Treatment Directive (1991), EU Water Framework Directive (2000)) measures to meet stormwater discharge and river water quality requirements have led to advancement in the design of urban drainage for diffuse pollution control. Sustainable Urban Drainage Systems (SuDS) are a range of urban drainage designs and new technologies (e.g. proprietary devices) that aims to replicate pre-urban development urban runoff (Fletcher et al. 2015).

The sediment transport through Sustainable urban Drainage Systems (SuDS) is complex. SuDS are best practice urban stormwater runoff and pollution management systems designed to convey the stormwater runoff from urban impervious areas by pipe infrastructure or environmentally engineered and overland flow paths into perennial (permanently flowing) and ephemeral detention and conveyance measures appropriate to flow attenuation and water treatment. SuDS comprise of a combination of features, including: perennial or ephemeral urban flow paths or channels, wet and dry vegetated water treatment measures, above and sub-surface drainage pathways and infiltration measures. The sediment transport process through this network is comprised of source (i.e. the urban surfaces on which sediment accumulated during periods of dry weather), pathway (i.e. the conveyance route) and sink (i.e. the receiving waterbody or the location where the sediment settles permanently) elements. To date, the design, monitoring and modelling of SuDS stormwater measures has been focused on specific rainfall-runoff events or flow discharge; for example, SuDS's role in water quantity relates to flood risk management where it is commonplace to design for a single flow (flood) hydrograph event (i.e. by reviewing local hydrology from source to sink for a very limited and specific 'event scale' time period (Fletcher et al. 2015, Woods Ballard et al. 2015). As a result of the design event focus, multiple event functionality of SuDS assets and networks has undergone limited analysis and there is uncertainty in sediment and pollutant (i.e. heavy metal) build up with SuDS assets. The critical literature review provided within this chapter intends to illustrate the current knowledge of SuDS sediment and pollutant detention/conveyance processes and highlight the research gaps from which the research presented in this thesis stems.

2.2 Urban sediment source, characteristics and loading rates

The influence of urbanisation on sediment production and conveyance within a catchment is diverse and complex; yet, it is possible to discern land-use correlations and wider runoff relationships which are generically applicable across catchments and appropriate to SuDS design modelling and guidance.

2.2.1 Urban development

Pizzuto et al. (2000) identified that urbanisation may result in positive and negative sediment production fluctuations, which can be evidenced from comparison of the 'open space' (i.e. undeveloped) data in Figure 2.1 to that of other land-use types. Typically, the catchment response will be an *increase* in sediment production due to urbanisation. For example, the small case study catchment (7.5km^2) of Pipers Creek, Washington, USA (Barton 2002) showed sediment production rates post-urbanisation to be six times greater than pre-development rates (tonnes/km^2/year). The average sediment discharge directly from urban development (90% of the catchment area), which was predominantly medium-density housing, was calculated to be approximately 28% of the loading. However, indirect processes of urbanisation relating to channel enlargement (due to both, the legacy of increased sediment loading during urban construction and the increasingly flashy hydrologic response to urban expansion of impervious surfaces) and channelization engineering (e.g. straightening) increased catchment erosion and conveyance processes as other primary sediment sources (~40% within this case study) persisting over the long-term.

Focussing on the physical construction phase of urban development shows that it can cause a significant increase in sediment production if inappropriate and/or ineffective sediment and erosion control measures are implemented. Indeed, this phase of urbanisation has specifically been found to particularly increase the *finer* sediment fractions in the sediment supply Particle Size Distribution (PSD) (Hubbart 2012, Taylor and Owens 2009). The literature-based evidence compiled and summarised in Figure 2.1 clearly shows construction sites to generate 100-10,000t/km^2/year; this is generally at least an order of magnitude greater than the 'matured' land use typologies post-construction i.e. a typical range of 10-100t/km^2/yr as demonstrated in Figure 2.1 (Haster and James 1994).

Figure 2.1 is an illustrated summary of available published field monitoring sediment load rates for urban surfaces to demonstrate the range and average sediment loading rates of urban areas. Figure 2.1 shows the mean and range of loading rates, in mg/L and ton/km^2/year taken from published literature. This provides a literary basis for sediment loading assumptions for generic urban areas and illustrates the range, and therefore uncertainty, in urban sediment loading specific to land use type.

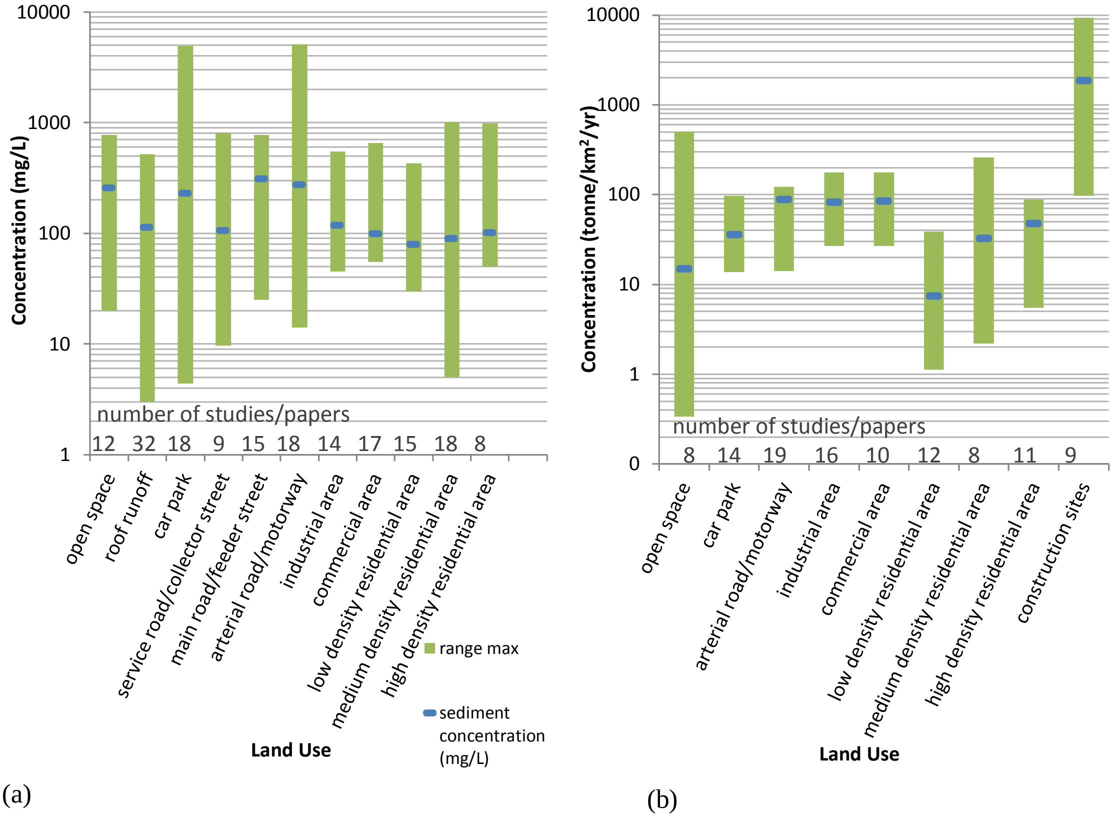

Figure 2.1 Land use specific sediment loading (concentrations) of urban surfaces. Figure 2.1(a) presented a summary of suspended material concentration in stormwater runoff from tested sources, Figure 2.1(b) presents reported urban surface mass sediment loading for monitored and published sources. Average values are indicated in blue with the reported range illustrated as the green bar. This data set comprises evidence complied from multiple publications (30 papers)[1]. The number of papers or studies included in each summary analysis are noted on the figures as 'number of studies/papers'.

For post-construction urban centres, the sediment loading (Figure 2.1) is highly variable according to specific land use types. As the Figure 2.1 comprises the full range of evidence available from the review work undertaken as part of this thesis, the detail of each (30 papers) study and land-use type is not provided herein; rather, only exemplar publications of relevance to this thesis are discussed. One such study is that from Taylor and Owens (2009), using data on the Aire-Calder catchment near Leeds, England (no significant SuDS implementation). This identified approximately 40% of sediment in

[1] Barton 2002, Crabtree et al. 2006, Deletic and Orr 2005, Droppo et al. 2010, Egodawatta et al. 2006, Ellis 1996, Erikssom et al. 2007, Faram et al. 2007, Gobel et al. 2007, Haster and James 1994, Hatt et al. 2004, Jartun et al. 2008, Jones et al. 2008, Lampe et al. 2004, Lau and Stenstrom 2005, Nelson and Booth 2002, Nie et al. 2008, Pal et al. 2011, Selbig and Bannerman 2011, Selbig et al. 2013, Spitzer and Jefferies 2007, STEPL, Taylor and Ownes 2009, Thorpe and Harrison 2008, Timperley et al 2005, Wang et al. 2013, Water by Design 2010, Wei et al. 2013, Whitehead and Crabtree 2007, Zanders 2005.

the urban river basin reach to result from road or sewer sources; given that the spatial coverage of these assets is disproportionately smaller than their contribution they are a highly significant source of sediment supply that can be locally managed by SuDS. This is reflected in the data of Figure 2.1, where transportation infrastructure demonstrates sediment loadings an order of magnitude greater than residential land-uses. Also, the same study shows that the associated increases in channelized flow path infrastructure (pipes, road gullies etc.) led to the deposition zones within the urban stormwater network being notably different from natural networks (Taylor and Owens 2009); this suggests that unique stormwater modelling and maintenance strategies be developed (rather than placing dependence upon existing sediment transport equations traditionally evolved from open-channel larger scale watercourse systems).

Analysis of urban density influence on sediment provision illustrates a persistent link between density and sediment loading. The increase in sediment loading from low to high density urban development (illustrated in Figure 2.1) is evidenced in case studies such as Reinelt (1996) (presented in Nelson and Booth 2002) and Egodawatta and Goonetilleke (2006) where surface sediment load collections from low to high density development increase from 70 to 600% (respectively) dependent on catchment and antecedent dry conditions. Thus, published studies illustrate (residential) development density to positively influence sediment loading (Egodawatta and Goonetilleke 2006).

Further case studies have identified that the sediment production rate post urbanisation is estimated to be six times greater than pre-development rates (tonnes/km^2/year) (Barton 2002).The average sediment discharge directly from urban development within the examined watershed was calculated to be approximately 28% (watershed urban extend of 90%) (Barton 2002). Channel enlargement, erosion and landslide were the other primary sediment sources (40% and 30% respectively) within this published case study. This examination illustrates that urbanisation contributes sediment directly but also causes change to channel and land stability, resulting in further increase to the waterway sediment load (Barton 2002).

2.2.2 Influence on supply characteristics

Whilst the study of Barton (2002; Section 2.2.1) suggests urbanisation to increase material <8mm, this combines analysis of sediment wash-off from urban surfaces with that derived from in-channel erosion. Reviewing the wider literature shows general acceptance that urban sediment has a Particle Size Distribution (PSD) skewed towards fine particle sizes, such that surface wash-off loads are predominantly within the bounds of silt and clay (Zander, 2005; Kayhanian et al. 2012). Droppo et al.'s (2010) field monitoring supports this hypothesis, suggesting the majority (up to 75%) of urban wash-off is below 63µm. Similar statistics are derived from other studies, suggesting that ~68% of wash-off load is <100 µm (Kayhanian et al. 2012, Li et al. 2006). From the >20 papers evidencing urban PSD, Figure 2.2 has been generated; this shows general consensus that urban sediment composition is weighted towards smaller sediment sizes in that >80% of urban material is <2mm (i.e. sand, silt or clay), independent of land-use source.

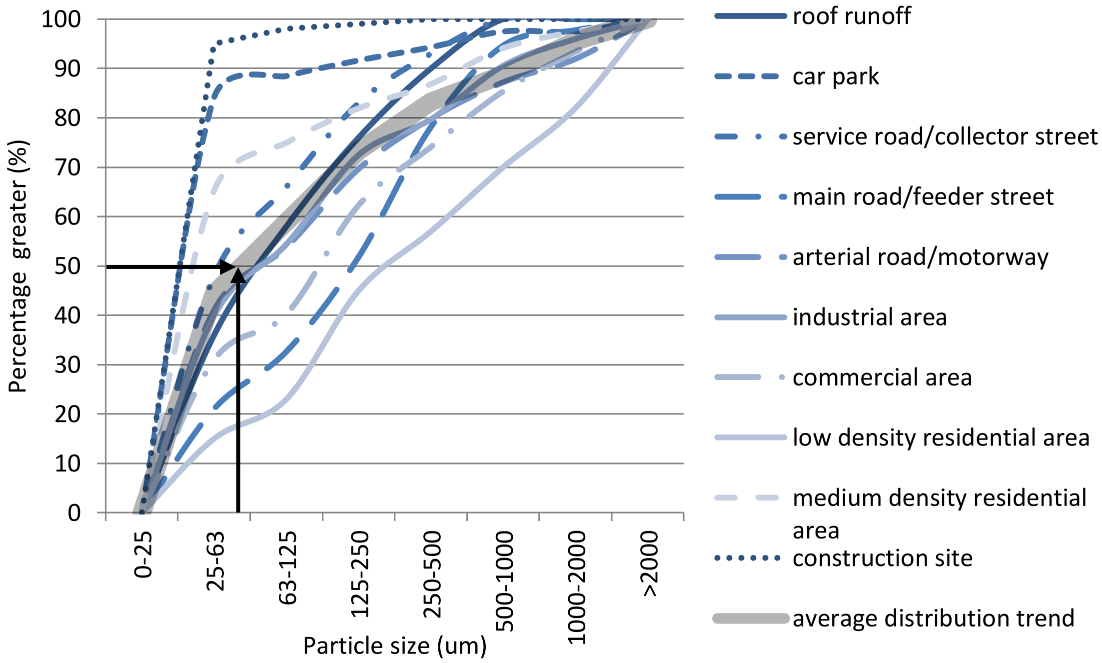

Figure 2.2 Summary of literature reported urban sediment particle distribution. This data set comprises evidence complied from multiple publications (25 papers), provided in the footnotes[2]. The arrows highlight the 50% or d_{50} particle size for the average distribution trend found across the publications.

[2] Barton 2002, Charlesworth and Lees 1999, Droppo et al. 2010, Ellis and Revitt 1982, Eriksson et al. 2005, Eriksson et al 2007, Haster and James 1994, Hubbard 2012, Jones et al. 2008, Kim and Sansalone 2008, Leibens 2001, Nie et al. 2008, Paul and Meyer 2001, Pizzuto et al. 2000, Roger et al. 1998, Sansalone et al. 1998, Selbig et al 2013, Sutherland 2003, Taylor and Ownes 2009, Timperley et al. 2005, Van Metre and Mahler 2003, Viklander 1998, Zanders 2005, Zhao et al. 2010, Zhu et al. 2008.

From Figure 2.2 it is evident that many studies concur that most land-uses provide a median grain size (d_{50}) of silt (~63µm) and d_{90} <2mm (grey line presented in Figure 2.2). However, construction sites and car parks repeatedly show finer PSDs of d_{50} around the clay-silt threshold of 25µm and the majority of the load (~d_{90}) being silts or clay. Conversely, main roads and low density housing have a coarser PSD (Figure 2.2) with d_{50} falling within fine sands (125-250µm); the bimodality of the road PSD reflect distinction in particulate loadings from vehicle (fines) versus road (coarse) abrasion (Adachi and Tainosho 2004, Thorpe and Harrison 2008). This is an important point, as published urban road pollutant analysis illustrates that it is the finer particulate modal group which exhibits the highest pollutant concentrations; e.g. Zanders (2005) states <500µm to show heavy metal pollutant concentrations in excess of coarser fractions, whilst Kayhanian et al. (2012) suggests 1000µm as the threshold (further considered in Section 2.3). This is logical, given that vehicle particulates will be non-sedimentary (e.g. metals, rubber, plastics); hence, Zanders (2005) highlights a related issue in that different material types (from different land-uses and locations) exhibit different particle densities which subtly affect entrainment and deposition process. For example, this data identifies a road surface specific density range of 2140-2390kg/m^3 stating distinction to three key past studies (Deletic 2001, Meyer et al. 1995, Stone and Marsalek 1996) which adopted more traditional particle densities akin to natural sedimentary geology of quartz 2600, 2700 and 2780 kg/m^3. The lower densities found in Zanders (2005) studies may demonstrate the influence of oil, grease (covering particles) and rubber particulates within the road samples, resulting in a lower sample density range.

Thus, the sediment detention and pollutant mitigation design of SuDS needs to incorporate consideration of detention mechanisms specifically for the finer particle size fraction of the urban sediment (wash-off) load. Critically, these fractions have slower settling velocity, are more readily held in suspended load for longer periods of time, may be cohesive and may be re-entrained by even small discharge events; this potentially leads to faster conveyance through the SuDS and a lower probability of detention for effective pollutant remediation, both of which must be considered in asset design. The land-use relationship to PSD (and associated pollutants; Section 2.3) therefore appears relevant to SuDS design and performance.

2.3 Urban heavy metal and mineral pollutants source, characteristics and loading rates

Section 2.2 raised the influence of particle size on pollutant transport, specifically relevant in urban environments for those heavy metals and nutrients documented in Table 2.1 (Zanders 2005, Van Metre and Mahler 2003, Hubbart 2012). Luker and Montague (1994) suggest that *"up to 85% of pollutants"* are carried, bound or adsorbed to sediment particulates rather than being transported independently (Whipple et al. 1983, Haster and James 1994). Other field studies have shown up to 90% of pollutants being conveyed as particulate matter (Nie et al 2008, Wei et.al 2013). This subtle distinction shows that the proportion of dissolved, isolated and sediment-bound pollutants is specific to local catchment characteristics. Likewise, the actual pollutant loading (concentrations) varies due to urban land use and source surface. However, it is important to note that where sediment carriers are the mode for pollutant transport, this is via their preferential adsorption onto fine sediment (<2mm; Figure 2.3).

The developed urban environment introduces a variety of heavy metals and minerals as pollutants into stormwater runoff. Urban pollutants result from road use, vehicles, residential, commercial and industrial activities and residential building materials; as these different sources produce different compositions of the chemicals, it provides insight into contributing zones and source-sink routing. Key road use and vehicular pollutants are copper (Cu), zinc (Zn), lead (Pb), and nickel (Ni) (Amrhein et al., 1992), while urban calcium (Ca) sources include road marking, concrete and building material degradation, building paint and grit/de-icer (Fay and Shi, 2012, Adachi and Tainosho 2004). Ca is also used as a detergent additive in engine oil (Monaci et al. 2000). Barium (Ba), tin (Sn) and manganese (Mn) are found in road dust and urban soils as a result of anthropogenic influence (Wang et al. 2005). Cu, Zn, and Mn are resultant from vehicle emissions and fuel leakage (Duan et al. 2012; Monaci et al 2000) while urban Ni, Cu and Zn sources include diesel and gas vehicles, with notable increase in urban diffuse pollution levels of Ni in heavily trafficked roads (Duan et al 2012). Pb has been removed from petrol, however previously contaminated soil in the urban environment continues to influence the urban diffuse pollution concentrations. Modern sources of urban Pb include industrial pollutant activities, house paint, lead flashing and roofing material. Ba and Zn have been attributed to tyre wear while Cu and Mn are recorded to occur from brake use and degradation (Monaci et al. 2000). A summary of urban pollutants and their sources is presented in Table 2.1, compiled from 18 published studies.

Table 2.1 Summary of urban pollution sources. This table is a summary of evidence presented in multiple publications (18 papers), provided in the footnotes[3].

Pollutant	Urban pollution source
K	fireworks, iron and steel production, domestic combustion, fertilisers
Pb	residual from leaded petrol, tire fillers, lubricants, motor wear, industrial activities, house paint, lead flashing and roofing material, manufacture of batteries and insecticides
Fe	abrasives (in vehicles), vehicle engine wear, urban infrastructure wear/degradation,
Cd	tyre abrasion, vehicle lubricants, industrial and incinerator emissions, insecticides
Cr	corrosion of vehicular parts, chrome plating, brake linings
Mn	emissions from alloy, steel and iron production, combustion of fossil fuels, combustion of fuel additives
Mg	vehicle exhaust, vehicle alloy metal bodywork (die castings), diesel emissions
Al	abrasives (in vehicles), vehicle bodywork, urban building fabrication, paint
Zn	tyre abrasion, vehicle lubricants, industrial and incinerator emissions, grease and paint manufacturing, road salt
Ca	brake pads and vehicle mechanical fillers, road marking, concrete and building material degradation, building paint and grit/de-icer, detergent additive in engine oil, road dust suppressant
Na	Cement, urban soil dust, road dust, fertilizer
Ba	manufactured materials: tiles, vehicle clutch and brake linings, rubber, brick, paint, glass
Cu	tyre abrasion, vehicle lubricants, industrial and incinerator emissions, radiators, bearings and brush wear in vehicles, a component in paint, used in tanning processes and plastics
Ni	corrosion of vehicular parts, chrome plating, asphalt paving, diesel combustion, brake usage and degradation, motor and braking fluids and coolants, fuel (diesel/petrol), electroplating (wastewater processes), food processing
Sn	fungicide, sewage sludge, urban waste (cans, food and beverage containers), coal ash, used in glass, paint and rubber making
P	urban green spaces (fertiliser, weed management), laundry detergent and cleaning fluids, surfactant use, on-site sewer systems, animal waste

[3] Adachi and Tainosho 2004, ATSDR 2007, Alloway 2012, Atiemo et al. 2011, Duan et al. 2012, Falbe 1987, Fay and Shi 2012, Howe et al 2005, Kobringer and Geinopolos 1984, Kundu and Stone 2015, McBride et al. 2012, Monaci et al. 2000, Pernigotti et al 2016, Pitt et al. 2004, Rodrigues-Seijo et al. 2015, Schueler and Shepp 1995, Thorpe and Harrison 2008, Wang et al. 2005

Of the urban diffuse source pollutants, hydrocarbons, PAH and several toxic metals have been labelled ‘persistent pollutant’ (Wilson et al. 2005). Persistent pollutants terminology is generally used to describe organic toxicants, but have been used ‘pre-flight’ by Wilsons et al. (2005) in the definition of non-organic pollutants of concern illustrating long term elevated occurrence in river sediment. Persistent pollutants are found, through watercourse sediment surveys, to be adsorbed to sediment and occurrent at elevated concentrations in urban waterways as a result of urban land use and development. Wilson et al. (2005) in their survey of Scottish urban waterways identified Zn, Ni, Pb, Cu, Cr and Cd as persistent pollutants. Persistent pollutants show negligible degradation, decay or change in speciation and can cause bioaccumulation (for example in river sediments) (Mattina et al 2003).

With further regard to source contribution, Auckland Regional Council research (Timperley et al. 2005) identified the ratios of roof : road : natural surface source loading for local industrial, commercial and high density residential areas for selected heavy metals and sediment. The significant contributing surfaces were noted to be roofs (Zn, Pb) and roads (Cu) at ratios ranging from 1:0.8 to 1:146 (Timperley et al. 2005). Thus, while all urban surfaces contribute to urban pollutant loading, the balance of greater contribution is towards road systems. Section 2.2.2 has already detailed Zanders (2005) road pollutant versus grain size study, but a more detailed summary of published urban sediment pollutant levels has been compiled in Figure 2.3 from 4 main field studies (Hubbard (2012), Timperley et al. (2005), Charlesworth et al. (2003) and Selbig et al (2013)). The data presented focus on road and car park dust; this is due to the low ratio of roof contribution and resulting unavailability of wider data for roof dust or building material sources.

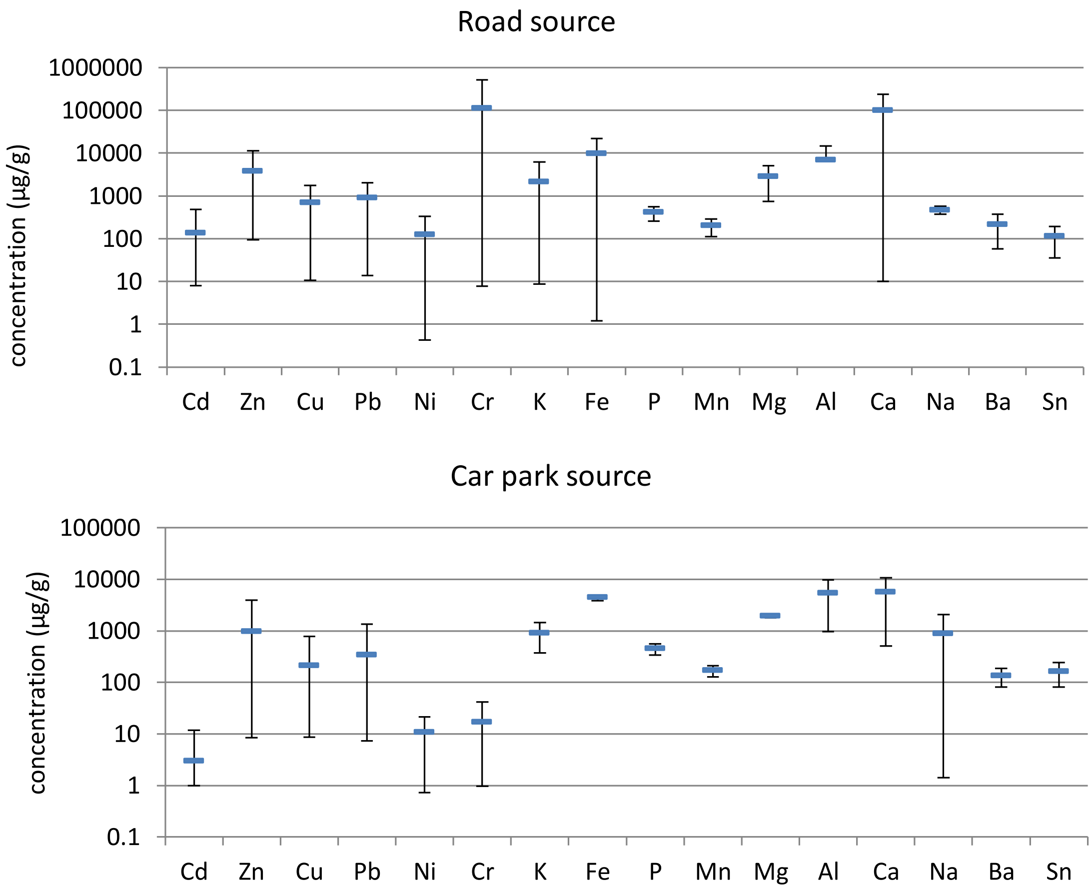

Figure 2.3 Chemical concentrations from urban sediment sources. The figures represent the summary of 39 publication[4] findings of urban pollutants, including Hubbard (2012), Timperley et al. (2005), Charlesworth et al. (2003) and Selbig et al (2013). Of these, 27 of the publications were relevant to road surfaces and 19 to car park surfaces.

Thus, Sections 2.2 and 2.3 provide an overview of the current known influences of urban sediment and sediment pollution. All reported pollutant concentration and particle size distributions for each land-use (i.e source) indicate a data range in Figure 2.3; hence, site-specific and local catchment influences must acknowledged for appropriate

[4]Adachi and Tainosho 2004, Alloway 2012, Arnato et al. 2009, Atiemo et al. 2011, Baudo et al. 1990, Calmano and Forstner 1993, Chapra 2008, Charlesworth and Lees 1999, Charlesworth et al. 2003, Deletic and Orr 2005, Ellis and Revitt 1982, Ellis 1986, Farkas et al. 2007, Heal 2000, Heal et al. 2006, Jartun et al. 2008, Kayhanian et al. 2012, Lau and Stenstrom 2005, Muschack 1990, Napier et al. 2008, Napier et al. 2007, Nie et al. 2008, Pal et al. 2011, Pitt et al. 2004, Poleto et al. 2009, Sansalone et al. 1998, Selbig et al. 2013, Sutherland 2003, Sutherland et al. 2012, Thorpe and Harrison 2008, Timperley et al. 2005, Van Meter and Mahler 2003, Vase et al. 2002, Victoria et al. 2014, Viklander 1998, Wang et al. 2013, Wang et al. 2005, Zanders 2005, Zhu et al. 2008

contextualisation and comparison of data analysis regarding urban sediment pollutants in a SuDS context.

2.4 A review of sediment transport processes through SuDS assets

2.4.1 Ephemeral and Perennial flow

Urban pollutants (sediment, minerals and metals) are conveyed off urban impervious surfaces by rainfall and runoff, flowing to and within SuDS as: (i) overland or piped flow for entry to the SuDS network; (ii) ephemeral or perennial flow for conveyance through SuDS assets. An understanding of the volume, rate and concentrations of all these flow transport processes is crucial to estimate and model the movement and detention of sediment pollutants within SuDS.

The process' differences between ephemeral and perennial flows with regards to sediment pollutant transport are the ease of entrainment and movement of sediment due to the wet/dry nature of the flow regime (Almedij and Diplas 2005, Reid and Laronne 1995). Ephemeral assets (e.g. swales, linear wetlands) have a stop-start flow regime directly influenced by rainfall and upstream discharge (Woods-Ballard et al. 2015, Melbourne Water 2005). Ephemeral flows show elevated stream power (ω)[5] (Knighton 1999), up to an order of magnitude greater than perennial flows of equivalent discharge and flow path (Almedij and Diplas 2005, Reid and Laronne 1995); theoretically, this results in greater sediment entrainment and conveyance during flow events. Conversely, perennial flows provide a more quasi-continual movement; although relative peak stream power is lower the increased water column depth prolongs the timeframe for particle suspension which increases transport duration along the flow path. While stream power analysis of ephemeral and perennial flows has been completed in river systems, these flow patterns (the dry/wet nature of flow) are known to occur in SuDS

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[5] Stream power is a numerical method to estimate the capacity of a flow to transport sediment (Bizzi and Lerner 2015). Moving water has a potential energy that can be transformed into kinetic energy (work) if the flow is channelised or contained within a defined cross section (Knighton 1999). Specific stream power follows the form (Kinghton 1999) $\omega = \tau_0$ and provided a numerical description of the energy supplied to the channel bed per unit area. The larger the energy provision to a specific channel bed area, the greater the potential for sediment transport. Thus, higher stream power values result in greater sediment transport (Knighton 1999, Bizzi and Lerner 2015).

and thus theory of greater sediment transport/erosion resultant from higher stream power in ephemeral flows may be transferable to SuDS sediment transport. In these assets finer sediment particles may be deposited on the bed of the flow path bed such that larger particles effectively sort, rearrange and “hide” particles from flow shear stresses of re-entrainment (Wilcock and Crowe 2003, Kleinhans and van Rijn 2002). As a result, the availability of fine material on a perennial flow path bed may be lower than that found in ephemeral assets (Almedij and Diplas 2005) such that fine material transport rates are far less in perennial systems.

2.4.2 Cohesive and non-cohesive sediment transport

Fine sediment, sediment (and organic) material of 63µm or smaller can be described as cohesive; having the potential to collide and form composite particulates that are larger than the individual properties (Droppo et al. 2004). Large sediments, coarse grain size (>63µm) in fluid generally act as individual particles, forming composite particulates only through mechanical interaction (i.e. in dense suspended solid concentrations) or addition of mechanical aids (McLean 1992). However silt and clay particles demonstrate a physio-chemical aggregation (flocculation) that is influenced by flow dynamics, environmental characteristics and organic matter (Droppo et al. 2004, Grabowski et al. 2011) and supports the formation of irregularly shaped, weakly bound particulates comprised of silt/clay sized particles.

Non-cohesive sediment follows the suspension, setting and bed load transport processes commonly discussed and employed in fluvial sediment modelling (e.g. HECRAS, SWMM and MIKE11 fluvial geomorphology modelling) (Merritt et al. 2003). Flow dynamics, such as velocity, shear stress, turbulence in conjunction with particle size and density form the key parameters in non-cohesive sediment transport equations (Ackers-White 1973; Engelund and Hansen 1967; Toffaletti 1969; Yang 1973). Cohesive sediment is more complex; flocculation occurs as a result of mechanical and physio-chemcial process, in response to electrostatic forces, environmental conditions including change in salinity, turbulence and temperature, and organic activity such as sludge or bacterial/microbial growth (Johansen 1998). As a result, cohesive sediment, when formed as flocculants (floc particulates) demonstrate a different settling velocity and therefore transport rate compared to their respective individual particles or a single particle of equivalent size. Mehta (1986) presented results of floc settling velocity relative to suspended sediment concentration (Figure 2.4) illustrating an increase in floc sediment settling velocity up to a threshold (>10kg/m^3) after which settling velocity declines.

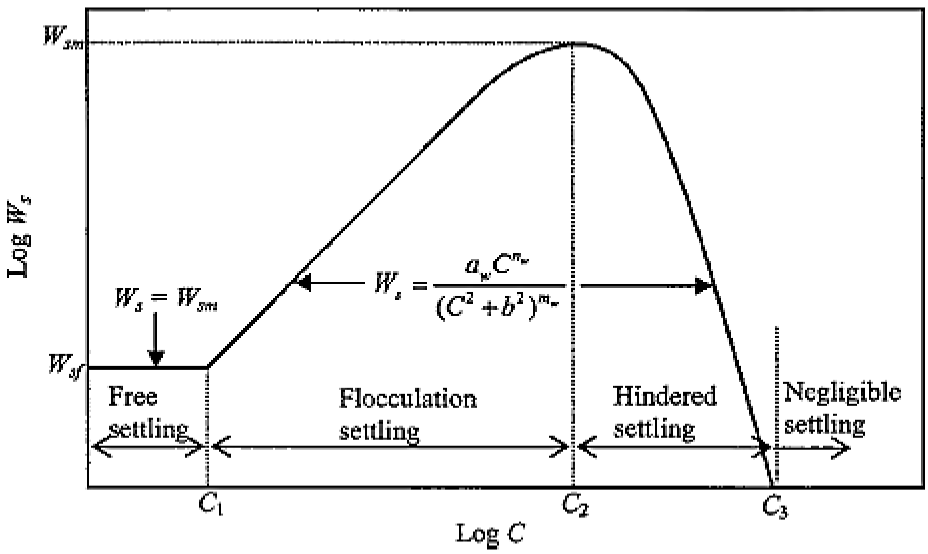

Figure 2.4 Reproduction of Mehta (1986) settling velocity versus suspended sediment concentration results, as presented in McAnally et al. 2007 (Figure 5, pg. 14). W_s is settling velocity, W_{s50} is free settling velocity, a_w, n_w, b_w and m_w are empirical settling coefficients and C is (total) fine sediment concentration.

The influence of cohesive sediment in transport analysis is in potentially elevated deposition of a greater quantity of finer sized particulates (material <63µm). Where individual <63µm *non*-cohesive particles would remain in suspension at a specified flow velocity and shear stress, flocculated sediment comprised of <63µm material may become deposited (illustrated by Mehta 1986 diagram, Fig 2.4). Vegetation filtering (physical detention of sediment due to flow path porosity) is also increased with the presence of floc particulates due to their increased particle size (Vastila 2010). Thus, with respect to SuDS (vegetated flow paths), the occurrence of cohesive sediment and therefore floc particulates would be expected to illustrate higher clay/silt detention relative to fine particle size (<63µm) comparative to non-cohesive sediment expectations (i.e. because of cohesive sediment formation of flocs, more fine sediment will become detained than would be expected if sediment was analysed by individual particle size), Furthermore, the creation (flocculation), stability and disaggregation of cohesive sediment floc particulates are influenced by temperature, salinity and turbulence, Settling velocity decreases with increasing temperature due to change in kinematic viscosity (Lau 1994, Naghipour et al. 2014). Saline water supresses the repulsive charges in cohesive sediment; as salinity increases flocculation potential increase and thus settling velocity (to the recognised threshold, <10kg/m^3) increases (Naghipour et al. 2014, McAnally et al. 2007). Low turbulence and shear stress can increase the potential for cohesive flocculation to occur, as turbulence increases it can result in the disaggregation of the weaker, irregularly shaped, floc particulates (Naghipour et al. 2014).

It is acknowledged that cohesive sediment, and its potential or capacity to form floc particulates, form an important element of urban sediment transport through SuDS. It may be expected that where elevated silt/clay (<63µm) is deposited despite moderate flow dynamics (velocity, shear stress, turbulence) cohesive sediment aggregation and transport processes may be influential. However, due to the complexity of analysis of cohesive sediment floc particulates in the field (requiring 2D imaging such as MRI or photogrammetry equipment) and the difficulty in maintaining field sample cohesive particulate stability for later analysis in the laboratory, identification and quantification of cohesive particulates was beyond the scope of this field study. It is acknowledged that this is a limitation of this study and that while undefined or quantified, cohesive sediment processes may assist in explanation of research findings and should form part of future research activities.

2.4.3 Dissolved, suspended and bed load transport

Urban pollutants can occur in adsorbed and dissolved forms. Section 2.2 notes that up to 85% of urban pollutants are adsorbed to fine sediment (Luker and Montague 1994). This solid load can be transported via bedload, suspension and/or saltation processes, depending on the relative balance of particle weight (i.e. size and density) to lift-drag forces of the applied fluid and flow velocity. The processes controlling dissolved pollution movement are advection, diffusion and dispersion (Wallis et al. 2009). A summary of the transport modes is provided below, in order to demonstrate context for empirical descriptors used in SuDS design equations and models.

Dissolved pollutants in solution move as part of the stormwater flow, at the same velocity (Vested et al. 1993). *Advection* processes describe the movement of a parcel of dissolved pollution without a change in the concentration of the pollutant; this process is also known as conservative mass transfer. *Dispersion* occurs when turbulence results in mixing of the dissolved pollutant with the surrounding waterbody; flow turbulence is a function of flow velocity, density and viscosity and can be numerically described using Reynolds numbers (Re) (van Rijn 1984). Due to the solute form of the dissolved pollutants, the driving forces behind dissolved pollution transport are therefore fluid flow conditions.

Adsorbed pollutants, attachment to the surface of sediment, require a far greater amount of fluid-derived energy to achieve transportation due to the inherent weight (i.e. resistance) of the particle. For a given stormwater flow, smaller (or lower density) particles can be entrained into *suspension* whilst larger (or denser) particles can be moved via *bedload* (i.e. rolling, sliding etc.). *Saltation* is the transitional process between these two transport states.

Suspended sediment processes relevant to SuDS include: transport, dispersion, deposition (flow velocity is lower than particle settling velocity), cohesion (in clay fractions) and physical detention via vegetation-based filtration (van Rijn 1984, Bagnold 1966). The determination of suspended sediment movement is based on particle density: fluid density ratio, flow velocity and particle settling (fall) velocity, discharge and turbulence, with some consideration of the filtering, drag and bed roughness factors of the conveyance channel (van Rijn 1984). There are multiple computational methods and equations available (Papanicolaou et al. 2008, Merritt et al. 2003) to define suspended sediment transport, considering a variety of particle size/density, hydraulic conditions and channels (further discussion of sediment transport equations within Section 2.5.4). Entrainment, particle settling velocity, decay rate (driven by particle size and settling velocity), diffusion and concentration of suspended sediment in are incorporated into dynamic models to calculate suspended sediment transport. Models such as MUSIC, SWMM, HecRas ST and MOUSE use buildup-washoff[6] and decay rate analysis (Section 2.5.3) to calculate sediment movement, while

[6] Build-up and wash off models (Shaw et al. 2010, Modugno et al. 2015) define surface sediment buildup through consideration of antecedent dry days, wind and vehicular influence on surface sediment deposition, impervious surface conditions and washoff through wind influence, rainfall and runoff

others (such as SWAT and ANSWERS) focus on the use of bed load or mass conservation equations to calculate sediment movement (Merritt et al. 2003).

Bed load sediment transport is the movement of larger or more dense sediment (and bound pollutants) though *saltation* (sediment bouncing along the bed) or *bed load transport* (rolling along the bed) (van Rijn 1984). The initiation of movement occurs when the drag and lift forces on the particle are greater than gravity and friction forces trying to hold the particle in place (Wilcock 1993), which can be defined by the critical shear stress (τ_c) (Shield 1936)[7]. Van Rijn (1984) provides a basic method of saltation distance estimation:

$$\frac{\lambda_b}{D} = 3D_*^{0.6}T^{0.9} \qquad \text{Eqn. 1}$$

Where λ_b is the saltation length, D is the particle diameter, D_* is the dimensionless particle parameter, and T is the transport stage parameter. To elucidate, all other things being equal, a smaller particle has a shorter saltation length than a larger particle at a similar transport stage. Material moving by saltation can travel further than bed load for a given flow and timeframe. Thus, separate bedload equations are required.

Over a Century of *bed load* equation research, development and revisions have been published, many of which combine rolling and saltating mechanics (Fernandez and Van Beek 1976, van Rijn 1984, Gomez and Church 1989, Reid and Laronne 1995, Wong and Parker 2006). Classic bed load equations (e.g. du Boys, Einstein and Bagnold equations) are typically based upon Shield's (1936) approach, using shear stress (critical and bed), flow velocity, particle diameter, gravitational acceleration, density (fluid and particle), depth (when incorporating saltation) in calculating bed load transport. Many bed load transport equations have been verified using larger particles (<200µm) and therefore may result in uncertainties and calculation difficulties when applied to smaller particle sizes available in SuDS (Gomez and Church 1989).

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duriation/intensity and loss through infiltration/leaching/degradation. The calculation of buildup and washoff is used to estimate stormwater runoff suspended solid and pollutant concentrations.

[7] Critical shear stress is the stress above which the gravity and friction forces will be lower than the drag and lift forces resulting in particle movement. Critical shear stress can be calculated using the equation: ${\tau_b}^* = {\tau_c}^*$, and $\tau^* = \frac{\tau}{(\rho_s - \rho_f)gD}$, (Buffington and Montgomery 1997).

2.5 A review of current sediment and pollutant transport concepts

Catchment balance modelling tools for sediment budgets can be described as numerical, empirical or conceptual approaches to transport, erosion and deposition estimation. Merritt et al. (2003) provides a comprehensive review of 17 commonly used fluvial sediment transport, water quality and erosion models [8] used in urban and mixed catchment analysis. These findings are generally supported by similar reviews completed by Elliott and Trowsdale (2007) and Wang et al. (2013). Across these reviews it is noted that there is little standardization of water quality (sediment or suspended solid) modelling; rather models are case study or region specific such that model use requires scientific understanding and judicious selection. The majority of sediment and pollutant models (not specific to SuDS design) are numerical, employing a mathematical description of relationships to calculate process change. The results of physically based modelling include an understanding of the parameter and variable interactions and the equations that support trends and process (Merritt et al. 2003).The high level, conceptual or total catchment level analysis predominantly focuses on sediment balance and capacity modelling while asset or site specific analysis is often 2D or 3D and process driven. Empirical models rely on and are created from verified observation while conceptual models are the representation by conceptualisation of a database, relationships and trends, providing a high level view rather than detailed semantics (Refsgaard and Henriksen 2004, Kandelous and Simunek 2010). To date, SuDS have been modelled conceptually (using, for example, MUSIC, SWMM), with little development of available fluvial sediment transport models appropriately revised to the complexities of SuDS flow path dynamics.

2.5.1 Mass balance analysis

Mass balance modelling can be considered in two frames (Tsakiris and Alexakis 2012), following the theory of conservation of mass; (i) pollutant transport, focused on suspended load only (movement of pollutants adsorbed to sediment in suspension); (ii) sediment transport, which focuses on bed and suspended load using Exner's equations (and advancements thereof) (Paola and Voller 2005, Tsakiris and Alexakis 2012).

[8] USLE, AGNPS, ANSWERS, CREAMS, EMSS, GUEST, HSPF, IHACRES-WQ, IQQM, LASCAM, LISEM, MIKE-11, PERFECT, SedNet, SWRRB/SWRRB-WQ, TOPOG, WEPP (Merritt 2003)

In its most basic form, the mass balance equation can be written as:

$$C_{out} = C_{in} + C_{lost} + C_{gained} \qquad \text{Eqn. 2}$$

where C is the mass of the element under consideration. Mass balance analysis is an accounting method that allows the form and location of a pollutant to be calculated at a specific point in time relative to previous hydraulic/hydrologic events over a defined preceding period (Chapra, 2008). When considered with regard to sediment transport, all hydraulic and morphodynamic processes may be incorporated into this equation to account for total sediment movement.

Mass balance is used most extensively in the analysis of pollutant transport, rather than for clean sediment (Jain et al. 1998, Sekhar and Umamahesh 2004, Warren et al. 2007). Mass balance pollutant models include software such as QWASI Model (Warren et al. 2007, Mackay et al. 1995), AARDVARK, QUAL2K, TOMCAT, SIMCAT, STREAMS, DRAINMOD, HecRas ST, Mike-11 (Ellis 1988, Tsakiris and Alexakis 2012)). Alternatively, where adsorbed or colloidal pollutants are considered, they are often modelled separately from solute pollutants (as TSS or sediment). Models such as MIKE-11 analyses sediment separately to solute pollutants; decay rate methods define solute pollutant movement, advection-dispersion considers cohesive sediments and sediment continuity equations model non-cohesive sediment (Neary et al. 2001). In models such as SIMCAT, MUSIC and SWIMM water quality analysis of TSS and solute pollutants are accounted for using continuous stirred-tank reactor (CSTR) differential mass balance analysis (CRCCH 2005, Cox 2003) while adsorbed (rather than solute) pollutants are not included in the water quality analysis.

Thus, for sediment modelling the earliest adopted conservation based sediment transport equation is Exner's equation, which uses continuity theory (Paola and Voller 2005). Exner's mass balance equation is defined as:

$$\frac{\partial \eta}{\partial t} = -\xi \frac{\partial U}{\partial x} \qquad \text{Eqn. 3}$$

Where η is the bed elevation above a selected reference datum, t is time, ξ is a coefficient, U is the average flow velocity and x is the distance downstream (Paola and Voller 2005). ξ can be defined as the porosity factor, calculated as 1/(1-porosity of the bed). This form of the Exner equation focuses on the conservation of bed sediment mass, rather than total load, thus describing bed load within a fluvial flow (Parker et al. 2000), and embedded principles of morphodynamic change such as transport via creation and movement of dunes. The underlying assumption of the Exner equation is that the rate of erosion and deposition is a function (proportional) to streamwise flow velocity (Ancey 2010).

Advances of the Exner's equation also incorporate suspended sediment, towards total load estimation and calculation of sediment flux in a selected river reach. The total load sediment flux form of the equation is (Grams et al. 2013):

$$\frac{\partial \eta}{\partial t} = -\frac{1}{(1-\lambda_p)}\left(\frac{\partial V_s}{\partial t} + \nabla \cdot Q_s\right) \qquad \text{Eqn. 4}$$

Where λ_p is the bed sediment porosity, V_s is the concentration of suspended sediment (volumetric) and Qs is the sediment flux. Many recent open-channel river sediment budget models have been undertaken using this form of mass balance analysis, particularly in conjunction with (St. Venant or Navier-Stokes) flow equations (Grams et al. 2013, Sinnakaudan et al 2003, Paola and Voller 2005, Aissiouene et al. 2016, Brunner 2010). Even for smaller scale fluvial processes, the Exner equation has been incorporated into more complex models to help consider bed sorting, initiation of movement and bed activity (Parker et al. 2000, Parker 1991, Lyn and Altinakar 2002, Wilcock and Crowe 2003) in fluvial (and in future potentially perennial) sediment transport.

2.5.2 Advection-dispersion modelling

Advection-dispersion modelling has been extensively used in coastal, fluvial and groundwater analysis to determine nutrient and pollutant movements. The physical characteristics of dynamic flow cause mixing and movement of nutrient/pollutant/substance in fluid (stormwater) resulting in the change in concentration and the volume of water conveying the substance. The substance velocity (in a specified direction) is not equal to the conveyance fluid resulting in dilution, detention, loss or potential intensification of the substance.

Advection-dispersion modelling simulates the movement of substances in a fluid (or gas). As such, the models focus on material in suspension. The 1D Advection-Dispersion Equation (ADE) is the integration of the advection and dispersion equations, and follows the form:

$$\frac{\partial C}{\partial t} + u\frac{\partial C}{\partial x} = D_L\frac{\partial^2 C}{\partial x^2} \qquad \text{Eqn. 5}$$

Where C is the substance concentration, t is time, x is the distance from the reference point and D_L is the dispersion coefficient (Parsaie and Haghiabi 2015). The dispersion coefficient is a function of fluid conditions, hydraulic conditions and the cross section geometry of the flow path.

ADE is commonplace in modelling (dissolved) pollution and suspended sediment transport e.g. models such as MIKE-11 and QUAL2E (Parsaie and Haghiabi 2015, Cox 2003). As ADE is effective in simulating non-reactive transport, water quality processes including kinetic reactions, decay rates, speciation and/or physical substance change are considered in parallel with ADE analysis, either through kinetic reaction analysis or chemical or physical source/availability analysis (Cox 2003, Kashefipour and Roshanfekr 2012). ADE modelling has been used successfully to simulate piped solute pollutants and suspended stormwater transport (Elliot and Trowsdale 2007) and, more recently, in SuDS ponds (West et al. 2016, Wallis et al. 2006, Spencer et al. 2011). However, a deficiency of these approaches are that the cohesive sediment and bed load transport processes are not incorporated into these ADE equations (Kashefipour and Roshanfekr 2012, Parsaie and Haghiabi 2015) and, hence, precludes full description of SuDS detention/conveyance sediment processes.

2.5.3 First order decay rate analysis

Commonly used industry-based SuDS models, such as MUSIC, SWMM and MicroDrainage adopt a different conceptual design approach. Here, the fine sediment transport does not follow classic river sediment transport descriptions (e.g. Ackers and White or Meyer-Peter Muller equations; Section 2.4.2 and 2.5.1); instead, water quality improvements include SuDS-based removal of suspended solids from urban surfaces by using first order kinetic decay rates.

Early water quality treatment measures focused on particle settling velocity and design flows to appropriately size detention and sedimentation basins:

$$A_s = 12\ Q/V_s \qquad \text{Eqn. 6}$$

Where A_s is the surface area of the basin, Q is the design flow or discharge through the basin and V_s is the selected particle size settling velocity (Ellis et al. 2004). This design relies upon the settling process to detain sediment and assumes that provided the SuDS asset is relatively large compared to the design flow passing through sediment will become deposited. The influence of vegetation, asset shape and dynamic flow are not incorporated. However, as early as 1953, research by Fair and Geyer (1953) presented a sediment removal efficiency equation incorporating asset shape into the basic particle settling velocity approach of water treatment. As shape influences flow path length, in turn, directly relating to residency timeframe and potential for remediation it was incorporated into the detention efficiency equation, together with related descriptors for flow turbulence (Ellis et al. 2004). The resultant sediment detention efficiency equation (R) therefore follows a settling tank functionality, leaning on waste water treatment design knowledge.

$$R = 1-[(1+1/n) \times (Vs/(Q/A))]^{-n} \qquad \text{Eqn. 7}$$

Where R is the sediment detention efficiency, Q is the design discharge flow, A is the SuDS asset surface area and n is the settling tank or continuously stirred reactor tank number (CRCCH 2005, Ellis et al. 2004). Here, ‘n’ is of specific importance; the smaller the value of n ($n \rightarrow 1.0$), the shorter the flow path and more turbulent the flow; hence, as $n \rightarrow \infty$ the flow path is seen to increase in length, flow turbulence overall is expected to decrease and sediment settling efficiency in expected to increase for a specified discharge. While the R equation guidance specifies values between 1 (very poor) to 5 (very good) (Ellis et al. 2004) when $n \rightarrow \infty$ the equation results in an exponential functionality with the detention efficiency becoming a function of time; in this case the R equation can be written as a time dependent exponential decay rate equation, taking the form:

$$R = 1-e^{-kt} \qquad \text{Eqn. 8}$$

Where k is the decay rate coefficient, calculated as Vs/h, h is the flow depth within the SuDS asset at the design flow rate (Q), and t is the residency time (V/Q) of sediment laden flow within the SuDS asset. Again, this equation is focused singularly on the design flow event and assumes that there is no shortcutting, turbulence or resuspension occurrence. This form of the R equation forms the foundation of current SuDS treatment measures that are designed using this efficiency computation approach to water quality analysis.

Thus, these first-order kinetic decay models are the currently accepted method by which to analyse pollutant removal and/or conveyance efficiencies for SuDS and blue-green drainage assets (e.g. MUSIC). The method employs a CSTR or plug flow assumption regarding pollutant transport and treatment (Woods-Ballard et al. 2007, Wong et al. 2002) such that the SuDS specific equation is described in Wong et al. (2002, 2006) to follow the form:

$$q\frac{dC}{dx} = -k(C - C^*) \qquad \text{Eqn. 9}$$

or, following the first-order kinetic decay equation format this can be rearranged as:

$$C_{out} = C^* + (C_{in} - C^*)\, e^{-k/q} \qquad \text{Eqn. 10}$$

where C = the concentration of the water quality parameter (mg/L); $q\ dC/dx$ = the rate pollutant concentration moves towards an equilibrium or background concentration with proportional distance along the treatment measure, C_{out}/C_{in}; C^* = the background concentration (mg/L); q = hydraulic loading rate (m/yr), the ratio of flow to surface area of the SuDS asset; x = the fraction of distance from the inlet to outlet; k = decay rate constant (m/yr) (Wong et al. 2006).

The first order kinetic decay rate computation, identified as the *k-C* equation*, has been used specific to sediment detention or conveyance, in the form of sediment transport capacity. In conjunction with MUSIC (CRCCH 2005) and SWMM (Rosseman and Huber 2016), sediment transport models such as WEPP have taken this approach to calculate the rate of particle deposition (Nearing et al. 2000) given that the first order decay rate coefficient (k) is defined as a function of sediment settling velocity (CRCCH

2005). When considered with an implicit assumption of direct proportionality between the sediment carrier concentration and associated pollutant concentration carried (Eslamian, 2014) the k-C* relationship can also be used to calculate pollutant conveyance efficiency. When taken one step further in explanation of relevance, as k is defined as a constant rate of change (Deletic 2001, Wong et al. 2006, Persson et al. 1999) it is possible to estimate the time taken for a pollutant concentration to change from its initial inflow concentration to the final attenuated, deposited and detained concentration (Newell et al. 2002). Thus, this equation acts to describe the overall movement of pollution from an event based pollutant influx to a longer-term equilibrium or background pollution level. It is therefore generally used and applicable in the description of not just the total suspended solids, but as also the nitrogen, phosphorus, biological oxygen demand and pollutant treatment efficiencies of SuDS (Wong et al. 2006).

The first order decay model is generally employed for steady state specific event analysis. Best practice guidance for k-C* modelling for SuDS design provides expected k constant values specific to expected or selected sediment particle sizes (CRCCH 2005). The guideline range of k values is from 4000 to 15,000 m/yr (CRCCH 2005), but could range across particle size specific k values presented in Figure 2.5 (based on data presented in Ellis et al. 2004, Table D1, p30).

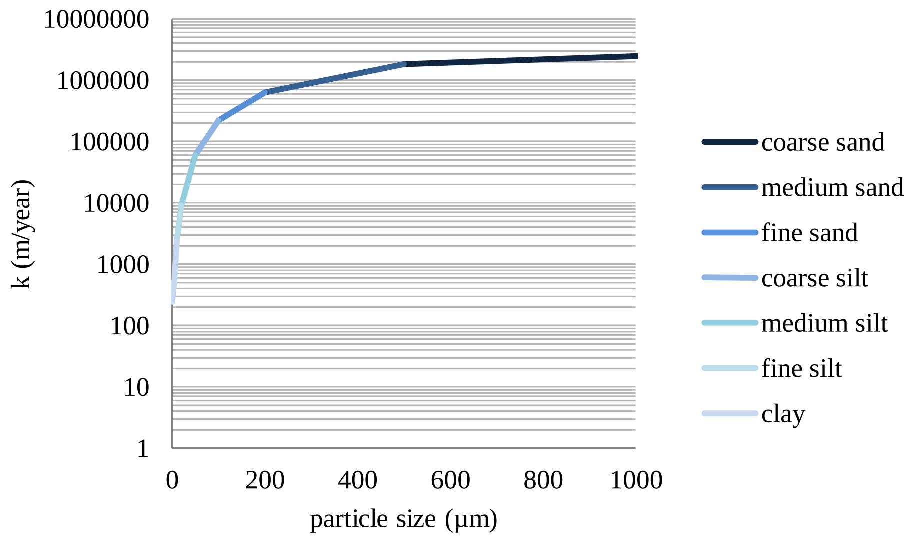


Figure 2.5 k values specific to urban sediment particle size. Figure 2.5 is a reconstruction of the k value guidance provided in the MUSIC model user's manual specific to the design and

modelling of sediment pollutant movement through SuDS (CRCCH 2005, Water by Design 2010).

2.5.4 Total sediment load transport

Sediment transport has been incorporated into hydraulic models to simulate the dynamic movement of sediment relative to flow. The equations used to describe sediment transport processes within these models can be characterised as shear stress relationship equations, discharge relationship equations or the probability of sediment movement relative to lift force (Munir 2011). Models such as e.g. HecRas, InfoWorks, FloodModeller, SWAT, MIKE-11, MOUSE, SWMM and SedNet incorporate sediment transport equations into their 1D modelling environment to simulate bed load and/or suspended sediment transport over specified time steps to be considered. A summary of four commonly used sediment transport equations utilised in industry adopted models (e.g. HecRas, FloodModeller) is provided in Table 2.2. Whilst these equations do accurately consider open-channel flow systems, they are typically design for channelized, perennial flows of faster velocity, greater unsteadiness and have limited analysis for very fine sediment (<63μm) (Dotto et al. 2010, Merritt et al. 2003, Brunner 2010, Gray and Simoes 2008, Haschenburgher and Curran 2012). In addition, they are less well tested in small channel cross-sections, nor where the flow path is densely vegetated; whilst some equations include a turbulence element (Re, Re_p) and/or roughness factor (Manning's n) to represent the influence of vegetation density it is unlikely to adequately reflect the very high density, diverse species (hence flexion range) and transitional emergent/submergent conditions of SuDS transport equation.

Table 2.2 Summary of four frequently used total load sediment transport equations and their key parameters in common sediment transport 1D models. It is noted that much or the research undertaken to create, describe and test these sediment transport equations result from laboratory based experiment rather than field work, due to the controlled nature of the laboratory environment. Appropriate sediment size is presented in millimetres.

Sediment transport function	Sediment transport equation	Notes	Bed load/ suspended load	Appropriate sediment size	Key parameters
Ackers-White (1973)	The sediment transport rate (G_{gr}), when A<F_{gr}) is: $$G_{gr} = C\left(\frac{F_{gr}}{A}-1\right)^{M}$$ where dimensionless particle size (D_{gr}) is: $$D_{gr} = d_s(\Delta g/v^2)^{1/3}$$ And $\Delta = \left(\frac{\rho_s}{\rho}\right) - 1$ the particle mobility F_{gr} is: $$F_{gr} = \frac{V_{.}^{*}}{\sqrt{gD(s-1)}}\left(\frac{V}{\sqrt{32}\log_{10}(10h/D)}\right)$$	Semi-empirical, energy based Effective for sub-critical flows Total load calculation is based on sediment mobility (F_{gr}), dimensionless grain size (D_{gr}) and the dimensionless transport rate (G_{gr}).	Total load	0.04-7.0	Hydraulic depth, velocity (V), flow depth (h), particle size (D), particle and fluid density (ρ, ρ_s) shear velocity (V*), viscosity (*v*), parameter describing initiation of motion (A)
Engelund	For total volumetric sediment transport:	Semi-empirical, energy based	Total load	0.15-0.93	Hydraulic radius (R), velocity (u), energy

Sediment transport function	Sediment transport equation	Notes	Bed load/ suspended load	Appropriate sediment size	Key parameters
Hansen (1967)	$G_{gr} = \frac{0.05wu^2h^{1.5}S^{1.5}}{\Delta^2 D\sqrt{g}}$ And for bed load : $q_t = 0.05u^2(RS)^{1.5}g^{-0.5}\Delta^{-2}D^{-1}$ Where $\Delta = \left(\frac{\rho_s}{\rho}\right) - 1$	Effective for sub-critical flows. Effective for larger sediment sizes. Assumes particle velocity is proportional to shear velocity and shear stress.			slope (water surface slope) (S), flow depth (h), particle size (D), flow/sediment density (ρ, ρ_s)
Toffaletti (1969)	$q_s = q_{ssL} + q_{ssM} + q_{ssU} + q_{sb}$ Where each sediment layer has a unique equation, such as: $q_{sb} = M(2d_m)^{1+Z_v-0.756Z_i}$ Where $M = 43.2C_L(1+Z_v)VR^{0.756z-Z_v}$ And $Z_v = 0.1198 + 0.00048(°F)$ And $Z_i = V\omega_i/(SRC_z)$	Theoretical, probabilistic An adaption and advancement of the Einstein equation (1950). Calculates both the suspended sediment load in the lower (q_{ssL}), mid (q_{ssM}) and upper (q_{ssU}) flow zones and sediment bed load (q_{sb}) separately to find total load (q_s).	Lower, middle, upper and bed load + suspended load	0.062-4.0	Flow velocity (V), flow depth (z), hydraulic radius (R), sediment fall velocity (ω), shear velocity (u*), viscosity (ν), temperature parameter (Z), sediment concentration (C), sediment size (d_m)

Sediment transport function	Sediment transport equation	Notes	Bed load/ suspended load	Appropriate sediment size	Key parameters
Yang (1973)	$$\log C_t = I_2 + J_2 \log(USP)$$ $$I_2 = 5.165 - 0.153 \log\left(\frac{\omega d_s}{v}\right) - 0.297 \log\left(\frac{u^*}{\omega}\right)$$ $$J_2 = 1.78 - 0.36 \log\left(\frac{\omega d_s}{v}\right) - 0.48 \log\left(\frac{u^*}{\omega}\right)$$ Incipient motion (for 1.2< Re_p>70) : $$\frac{V_c}{\omega} = \frac{2.5}{\log(Re_p) - 0.06} + 0.66$$ And effective unit stream power (USP) is: $$USP = \left(\frac{VS}{\omega}\right) - \left(\frac{V_v S}{\omega}\right)$$	Theoretical, energy based Effective for sub-critical flows. Total sediment concentration (C_t) shows low sensitivity to flow depth, velocity derivations (Fr, τ, ω) and sediment concentration. Effective for larger sediment sizes.	Total load	0.062-7.0	Unit stream power (USP), particle settling velocity (ω), Reynolds particle number (Re_p), viscosity, particle size (d_s), flow velocity (v)

(Dotto et al. 2010, Merritt et al. 2003, Brunner 2010, Gray and Simoes 2008, Haschenburgher and Curran 2012)

Table 2.2 illustrates that sediment transport equations are complex, often compartmentalised by sediment size (such as Toffaletti 1969) and incorporate numerous key flow, sediment and channel parameters within the equations. The most prominent parameters are flow velocity (or derivatives there of: Fr, Re), flow depth, shear stress (or alternative energy parameter such as shear velocity, energy slope) and particle size characteristics (diameter, density). These equations have been designed and tested for sand-gravel particle sized, with limited analysis and verification undertaken at fine sediment or cohesive sediment (<63μm) level (Dotto et al. 2010, Merritt et al. 2003, Brunner 2010, Gray and Simoes 2008, Haschenburgher and Curran 2012). Sediment transport modelling using equations such as those presented in Table 2.2 are effective in simulating natural channel sediment transport in river systems. They are less well tested in small waterways where the flow path is densely vegetated. While some of the common total load sediment transport equations include a turbulence element (Re, Re_p) roughness factor, the influence of vegetation density in the flow path is not always integral to the transport equation. While conceptual SuDS models, such as MUSIC and SWMM, are generally steady state, through analysis of SuDS sediment transport using the key total load parameters the multiple event influence of flow on SuDS sediment may be further explained (examined in Chapter 5).

To account for vegetation influence, recent research has applied turbulence equations (i.e. k-ϵ turbulence model, 2D depth-averaged model) to calculate the flow (velocity and discharge) and turbulence influence of vegetation on suspended and bed load sediment transport (Lopez and Garcia 1998, Wu et al. 2005).

Vegetation can influence bed shear stress, streamwise momentum transfer and residency time within a vegetated flow path (Lopez and Garcia 1998, Jordanova and James, 2003, Sonnenwald et al. 2016). Laboratory and field results have illustrated that vegetation influence in the flow path results in lower suspended sediment conveyance and bed load transport due to the reduction of energy (momentum and velocity), stem drag and reduced flow path porosity (Jordanova and James, 2003, Sonnenwald et al. 2016, Wu et al. 2005). The recent consideration of vegetation influence on sediment conveyance illustrates the importance of turbulence influence of sediment transport and the complexity of including this parameter into sediment transport equations. This may be of importance in the future analysis of vegetated perennial/ephemeral flow paths (e.g.

SuDS) and suggests that future research consider turbulence (such as Reynolds values) when considering sediment transport through these vegetated flow paths.

2.5.5 Sediment transport capacity

It is important to note that the sediment transport equations discussed in this section generally function on an equilibrium (unlimited) supply assumption (Brunner 2010, Ali et al. 2012, Yager et al. 2012). When modelling sediment transport, there is often an implicit assumption regarding the upstream sediment supply; that is that sediment transport occurs at sediment transport capacity. Sediment transport capacity can be defined as the total possible sediment able to be conveyed by a given flow (Huang et al. 1999). Thus the sediment transport rates calculated using the equations in Table 2.2 provide the maximum possible sediment conveyance for the flow under consideration rather than a sediment supply specific result. In the research undertaken within this thesis this assumption is not accurate for tagged sediment transport analysis. The quantity of tagged sediment released is limited, and thus while the total sediment movement may be in accordance with total conveyance capacity assumptions; the tagged sediment movement is supply limited. Therefore, it is important to note what sediment conveyance capacity is, and the parameters that control the conveyance capacity.

2.6 Current understanding and expectations of SuDS functionality

Sustainable urban Drainage Systems have been implemented in the UK, Australia, New Zealand, USA, Canada and more recently in areas of China, Japan and Singapore over the past two decades. Numerable studies on the event specific ‘functionality’ of SuDS have been undertaken and published either as academic literature or development and design guidance; herein the ‘functionality’ is defined/considered as the achievement of expected sediment and pollutant reduction as discussed by relevance guidance or legislation. The following Section provides a summary of the current published SuDS stormwater sediment and pollution treatment functionality identified through field and laboratory testing in conjunction with international guidance expectations of what effectively designed, implemented and maintained SuDS can or should achieve. This review has focused on four SuDS assets: wetland, linear wetland, swale and pond, as these are the field SuDS assets monitored and analysed in this thesis research. Other common SuDS assets, e.g. permeable pavements, filter strips, green roofs and infiltration systems are, therefore, not specifically considered herein.

The general premise on which SuDS are designed is the use of flow velocity reduction, temporary flow volume detention and sediment settling (Woods Ballard et al. 2015). Stormwater runoff from impervious urban areas is collected and detained within SuDS assets to: (i) delay the flow; i.e. to reduce the outflow rate and concentration to natural, undeveloped levels; (ii) attenuate the discharge, i.e. to increase stormwater storage in the asset so the outflow volume occurs over longer timeframes, with lower peak discharges and without detrimental impact on downstream flood risk; (iii) decrease the velocity of stormwater discharge, i.e.to minimise damage from scour/erosion/avulsion processes to the asset, outflow design and local watercourse; ; and, (iv) allow infiltration, i.e.to reduce the quantity of stormwater discharge. It is crucial to clarify here that the primary design objective for SuDS assets is flow control, with water quality improvement often an equal or secondary objective (Woods Ballard et al. 2015, Kouvelis and Armstrong 2004, Tetra Tech 2010, Healthy Waterways 2006, Shutes and Raggatt 2010, Hoyer et al 2011); this prioritisation of function is a common source of confusion of environmental and wastewater engineers.

Water quality improvement provision by SuDS is generally reliant on adsorption and settling of suspended and entrained urban sediment pollution. Through temporary detention of stormwater in SuDS slower flow velocities (comparative uncontrolled runoff/flow events) allow larger sediment to settle and become (permanently) deposited on the SuDS asset bed. Complimentary to this design, phytoremediation mechanisms are incorporated in SuDS. In definition, phytoremediation is the direct use of living green plants for in-situ removal, degradation or containment of contaminants via sludges, sediments or water bodies. Submergent and emergent vegetation all increase flow resistance (Manning's "n" values), slowing water movement local to the plant(s); this reduces the local kinetic energy of the waterbody, encouraging particle settlement and detention of sediment and associated pollutants. Once deposited within densely vegetated flow paths, the heavy metals and minerals can be remediated via plant processes including breakdown, degradation, fixation or a change in speciation (Salt et al. 1998, McIntyre 2003). Thus the design (e.g. size, depth, flow path length, roughness, vegetation density) potentially influences the degree of water quality improvement that is possible by a specific SuDS asset.

Interestingly, SuDS design efficiency standards differ across the world. Within the UK there is a hesitancy to legislate a prescribed efficiency value or threshold. Rather than stating a specific percentage value or threshold requirement for pollutant removal or design water quality concentration, the UK Local Authorities and national agencies state that commercial, industrial and major residential developments (over a specified size or density, ostensibly 10 residential properties (SI 2015/595, SPP1) development require SuDS treatment trains comprising 1, 2 or 3 SuDS assets (i.e. a SuDS network, Woods-Ballard et al. 2015). That said, the SuDS Manual (CIRIA, 2015) does provide indicative information on possible or expected SuDS efficiencies in both concentration (Table 2.3) and percentage reduction (presented in Tables 2.4-2.7 and derived from a comparison of Annex 1 and 3 of the SuDS Manual (CIRIA 2015)) (Woods-Ballard et al. 2015). The SuDS Manual is not, however, prescriptive (legislated) and therefore provides guidance to Local Authorities who can choose to adopt the guidelines. It appears that the reticence of the UK to prescribe values or thresholds arise due to the limited field data by which to inform water quality design standards, geographical variability of available data, poor temporal resolution of data and most notably the lack of long term field monitoring of SuDS to inform maintenance requirements (O'Sullivan et al. 2008, Williams et al. 2011). This lack of evidence is something that has been partly resolved in the USA via development of a large, pan-regional datasets of stormwater best management practice which is maintained and continually updated by the US EPA, WERF and other government agencies, with the support of external consultancies (Leisenring et al. 2014). This database informs the design expectations of SuDS (or Low Impact Development (LID)) assets across the USA and their findings inform the pollutant removal efficiencies stated in State or local stormwater management design guidelines. Whilst this large-scale evidence based approach remains an aspirational intention for the UK, the Australian WSUD guidance approach is largely equivalent to CIRIA SuDS manual advice. Arguably, Australian research has a larger field data set, lower regional variation in data and longer history of SuDS related modelling than the UK, yet they take a similar approach to SuDS design in terms of aiming to address water quality improvement or treatment to a specified level. Whilst the efficiency of individual assets is not defined specifically, the efficiency of the treatment train (regardless of the composition of network assets) is specified in terms of compliance with permitted discharge levels (Melbourne Water 2015, Healthy Waterways 2006, CSIRO 2006).

Table 2.3 UK SuDS Manual reported possible outflow pollutant concentrations from SuDS. These are the only pollutants discussed in the Manual (extracted from Annex 3, CIRIA 2015).

SuDS asset	Pollutant Effluent concentration				
	TSS (mg/l)	Cd (µg/l)	Cu (µg/l)	Zn (µg/l)	Ni (µg/l)
Wetland	4-21	0.1-0.4	2-6	11-33	-
Linear wetland (bioretention)	5-20	0.04-0.1	4-10	5-29	3-8
Swale	10-43	0.2-0.3	4-15	18-55	2-5
Pond	10-47	0.1-0.4	2-12	6-58	2-4

An overview of UK and international expected and reported SuDS performance is provided in the following sections. It should be noted that the literature published water quality treatment results summarised in Tables 2.4-2.7 are from single event (event mean concentration) sampling analysis. There remain few long term sampling studies; those available include e.g. Lambs Drove, Dunfermline Expansion (DEX) and other ponds across Scotland (Heal et al. 2000, CIRIA 2009, Heal et al. 2006) as well as overseas (e.g. Deletic et al. 2001, Hossain et al. 2005, Parker 2010 and Magette et al 1989). The monitoring across the SuDS network at Lambs Drove (UK) provided (by-monthly event based) suspended sediment and surface water quality results for stormwater runoff/flow through swales, filter strips, detention basins and a retention pond (Ogunyoye and Stevens 2012). While providing a detailed analysis of suspended and solute water quality improvement by the implemented SuDS assets, the rate of sediment deposition within these assets, and deposited sediment contamination, was not a focus of this study. Monitoring of Scottish ponds and wetlands: DEX (Heal et al. 2006); Caw Burn wetland (Heal et al. 2005); Dulock Park (Heal and Drain 2003), has identified sediment deposition (sampled annually) in ponds/wetlands and the sediment pollutant concentration of seven (Cd, Cr, Cu, Fe, Ni, Pb and Zn) metals. Source-pathway-sink analysis of pollutant movement, specifically sediment adsorbed pollutants, is not included in the findings of Tables 2.4-2.7 due to the lack of published data on multiple-event sediment and pollution movement, deposition or detention.

2.6.1 Wetland

A SuDS wetland is a constructed wetland, a continuously wet asset that has variable depths, vegetation densities and perambulating flow path which extends flow residency time (Healthy Waterways 2006, Woods Ballard et al. 2015). They are designed to both, provide sediment and pollutant removal for low flows, for example the 2 year RP flow event and, support flow control for major flood events i.e. 200 year RP in Scotland

(Woods-Ballard et al. 2015). Figure 2.6 presents a sketch of a constructed wetland and photograph of the J4M8 wetland.

(a) (b)

Figure 2.6 SuDS wetland illustration. Figure (a) is a photograph of the J4M8 wetland in the Central Belt of Scotland; this is a mature wetland constructed during the distribution park development (2001-2006) adjacent to a commercial industrial centre where units share moderate-sized communal car park space. Figure (b) is a design schematic of a constructed wetland taken from WSUD technical design guidelines (2006).

Constructed wetlands have been monitored to identify flow control and water quality treatment efficiencies internationally; Table 2.4 summarises the data from 13 studies across Europe, North America and Australasia. In many cases, pollutants illustrate a wider range of removal capability by wetland assets than that expected/required by design guidelines. There is evidence that the guidelines can be met; however local variability in e.g. environmental controls, land-use supply and/or asset design does indicate that sub-optimal performance is common. In addition, Table 2.4 clearly evidences the restricted number of pollutants for which guidance exists; which is surprising in light of the knowledge-base of wider urban pollutants, as demonstrated in Section 2.3. Similarly, variability in international guidance levels is evident, showing UK expectations or reported achievement of SuDS water treatment to be among the highest in the world. Compared to other countries' guidance (and the reported data of Table 2.4), it appears reasonable to question whether UK expectations or reported achievement is over-optimistic compared to reported wetland performance.

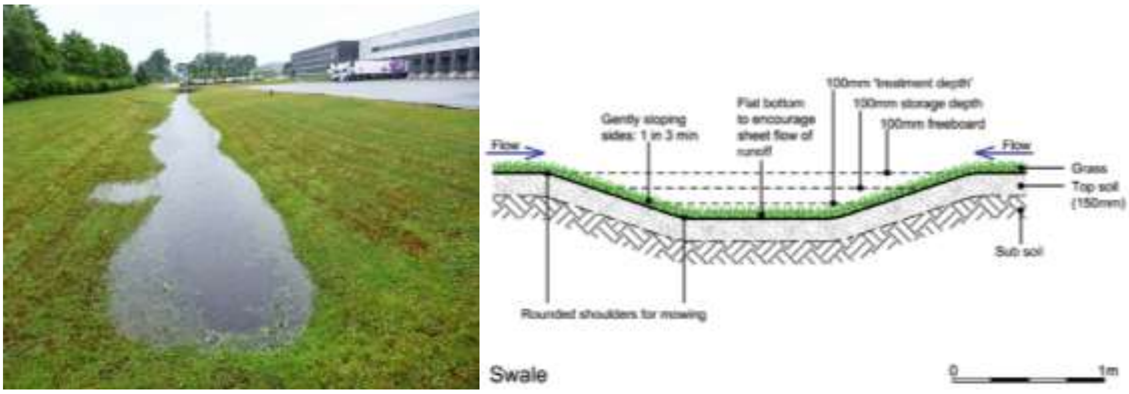
Table 2.4 Published wetland water quality and pollution removal levels (in %). Reported data are sourced from 13 international data sets; data ranges suggest multiple studies or repeat monitoring data are available, whilst single datum are from an individual field sample.

Pollutant	Reported removal rate %	Reported or expected removal efficiency rate %		
		UK[1]	USA[2]	AUS/NZ[3]
TSS	9-96	90-92	75	85
Zn	50-99	80-91	40-45	-
Cu	30-83	82-93	40	-
Total N	16-99	-	30	45
Total P	6-99	-	49	65
Pb	50-83	-	68	-
K	5	-	-	-
Ni	39-50	-	-	-
Ca	34	-	-	-
Mg	5	-	-	-
Na	34	-	-	-
Cd	80-92	67-82	-	-
Cr	2-89	-	-	-
Fe	74	-	-	-
Ba	-	-	-	-
Sn	25-50	-	-	-
Mn	-	-	-	-
Al	63	-	-	-

References: Maine et al. 2007, Maine et al. 2009, Khan et al. 2009, Brydon et al. 2006, Yeh et al. 2008, Nelson et al. 2006, Jayaweera et al. 2007, Diaz et al. 2012, Vymazal 2013, Huang et al. 2000, Clary et al. 2011, Kropfelova et al. 2009, Barret 2008,
[1] SuDS Manual (CIRIA 2015)
[2] LID technical guidance manual for Puget Sound (Hinman 2012), US EPA Stormwater best management practice design guide (Clar et al. 2004)
[3] Best Practice Environmental Management Guidelines (CSIRO 2006), Melbourne Water Constructed Wetland design guidelines (Melbourne Water 2015), DERM 2010.

2.6.2 Swale

A swale is a vegetated surface drainage path, designed to collect surface runoff and convey it downstream in a controlled manner. Swales are designed to control stormwater via both the delay of peak discharge conveyance downstream (due to shallow gradient) and the increase in flow path roughness due to vegetation (Woods-Ballard et al. 2015, Healthy Waterways 2006, Melbourne Water 2005). If unlined, swales can also support infiltration for a reduction of both flow volume and pollutant conveyance downstream. Swales are often located parallel to roads and car parks, and are planted with short, easily maintainable (e.g. standard mowing practices) grass species. Provision of grass coverage throughout the swale (central flow path and banks) also helps to detain stormwater pollutants (via physical detention and phytoremediation). Check dams (small dams placed within the swale to temporarily detain stormwater flow and decrease flow velocity (Yu et al. 2001) may be used within the swale flow path to further aid all processes (Woods-Ballard et al. 2015, Healthy Waterways 2006, Melbourne Water 2005). Figure 2.7 presents an illustration of a UK swale (at J4M8 field site) and schematic diagram of best practice design.



(a) (b)

Figure 2.7 SuDS swale illustration. Figure 2.6(a) illustrates an established grassed swale located within the J4M8 distribution park, Scotland. Figure 2.6(b) is a schematic cross section of an indicative grass swale, modified from the from WSUD technical design guidelines (2006).

Table 2.5 summarises the data from 12 field monitoring studies, demonstrating the actual versus possible design-expected improvements of swales for water quality (Woods-Ballard et al. 2015, Healthy Waterways 2006, Melbourne Water 2005, Clar et al. 2004). There is broad similarity in the UK, USA and Australasian design guidelines, generally towards the upper end of the reported field data range. Instances of swales being poor water treatment assets appear common, with low (<20%) reported removals rates recorded for many pollutant types.

Table 2.5 Published swale water quality and pollution removal levels (in %). Reported data are sourced from 12 international data sets; data ranges suggest multiple studies or repeat monitoring data are available, whilst single datum are from an individual field sample.

Pollutant	Reported removal rate %	Reported or expected removal efficiency rate in %		
		UK[1]	USA[2]	AUS/NZ[3]
TSS	10-95	76-82	80-84	85
Zn	2-96	60-95	71	-
Cu	7-94 (<25% dissolved)	60-84	51	-
Total N	10-67	50	45-84	45
Total P	12-65	55	34-45	65
Pb	17-95	-	67	-
K	3-51	-	-	-
Ni	40-78	57-84	-	-
Ca	-	-	-	-
Mg	13-74	-	-	-
Na	-	-	-	-
Cd	0-55	33-86	-	-
Cr	11-99	-	-	-
Fe	9-89	-	-	-
Ba	-	-	-	-
Sn	-	-	-	-
Mn	-	-	-	-
Al	-	-	-	-

References: Yousef et al. 1987, Rushton 2001, Istenic 2012, Stagge et al 2012, Wang et al 1980, Barrett et al 1998, Caltrans 2004, Backstrom 2003, MacDonald 2003, Shutes and Raggatt 2010, Clary et al. 2011, Barret 2008

[1] SuDS Manual (CIRIA 2015)

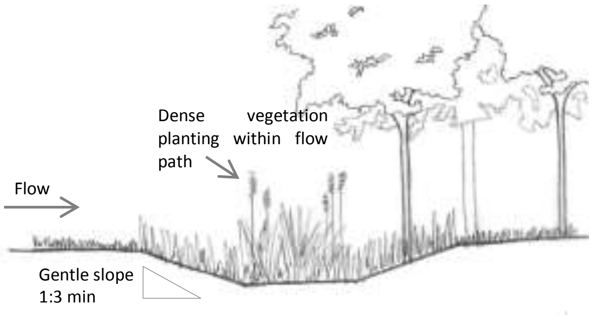
[2] Ontario Ministry of Environment(),LID stormwater management planning and design guide for CVC (Dhalla and Zimmer 2010),LID technical guidance manual for Puget Sound (Hinman 2012), US EPA Stormwater best management practice design guide (Clar et al. 2004)

[3] Best Practice Environmental Management Guidelines (CSIRO 2006), Melbourne Water WSUD engineering procedures (Melbourne Water 2005), DERM 2010.

2.6.3 Linear wetland

This asset is a variant of a swale. It's distinction is that the denser vegetation planting within the flow path is purposefully designed to more substantially detain and treat stormwater (rather than convey); in this case it is identified as a 'linear wetland' or a 'linear wet swale' in the UK (Woods-Ballard et.al 2015). Whilst still ephemeral in flow, it is wet for longer periods of time due to the enhanced detention design. In the USA and Australasia similar SuDS assets are termed as 'bioswales' and are often enhanced by subsurface drainage (gravel infiltration media sometimes provided with a subsurface collection and conveyance pipe) (Healthy Waterways 2006, Melbourne Water 2005, Dorman et al. 2013). Figure 2.8 provides a basic schematic of a UK linear wetland (photograph: J4M8 linear wetland).

(a)



(b)

Figure 2.8 SuDS linear wetland illustration. Figure 2.7(a) illustrates the established linear wetland located within the J4M8 distribution park, Scotland, located adjacent to the KNL commercial car park. Figure 2.7(b) is a schematic cross section of an indicative linear wetland (bioretention swale), modified from the from WSUD technical design guidelines (2006).

Linear wetlands are densely planted with water and pollutant tolerant species. The recommended planting varies according to location and climate but often includes hyperaccumulator species (Health Waterways 2006); these are plants capable of absorbing via their roots 2-3 orders of magnitude of heavy metals compared to normal plants and concentrating these extremely high levels of metals in their tissues for remediation by e.g. phytoextraction. Planting type is, therefore, a significant control on the water quality treatment provided by a linear wetland, yet it is also strongly influenced on the provision of infiltration measures, i.e. whether the design includes bioretention or infiltration medium. It is these design variations which lead to asset performance data demonstrating a very wide range in the 13 literature-based data sets summarised in Table 2.6. It is evident from this data that the UK reported possible values illustrate significantly greater performance efficiencies of these assets in metal remediation than those expected internationally; given the variability of monitored data UK SuDS Manual reported values, if used as a guideline of expectations, may be unrealistic or over-optimistic continually over the medium to long term treatment horizons.

Table 2.6 Published linear wetland (representative bioretention swales where appropriate) water quality and pollution removal levels (in %). Reported data are sourced from 13 international data sets; data ranges suggest multiple studies or repeat monitoring data are available, whilst single datum are from an individual published field sample.

Pollutant	Removal rate %	Reported or expected removal efficiency rate %		
		UK[1]	USA[2]	AUS/NZ[3]
TSS	3-92	90-92	70	85
Zn	25-98	90-92	71	-
Cu	4-99	60-89	51	-
Total N	13-89	50	84	45
Total P	0-90	80	34	65
Pb	53-98	90	67	-
K	-	-	-	-
Ni	56-90	35-74	-	-
Ca	-	-	-	-
Mg	-	-	-	-
Na	-	-	-	-
Cd	40-90	87-95	-	-
Cr	0-35	-	-	-
Fe	0-87	-	-	-
Ba	-	-	-	-
Sn	-	-	-	-
Mn	-	-	-	-
Al	63	-	-	-

References: Davis et al. 2001, Davis et al. 2003, Hunt 2003, Sun and Davis 2007, Hatt et al. 2007, USEPA 2000, Shutes and Raggatt 2010, Baudo et al. 1990, Hunt et al. 2006, Hsieh and Davis 2005, Jurries (ed) 2003, Clary et al. 2011, Barret 2008

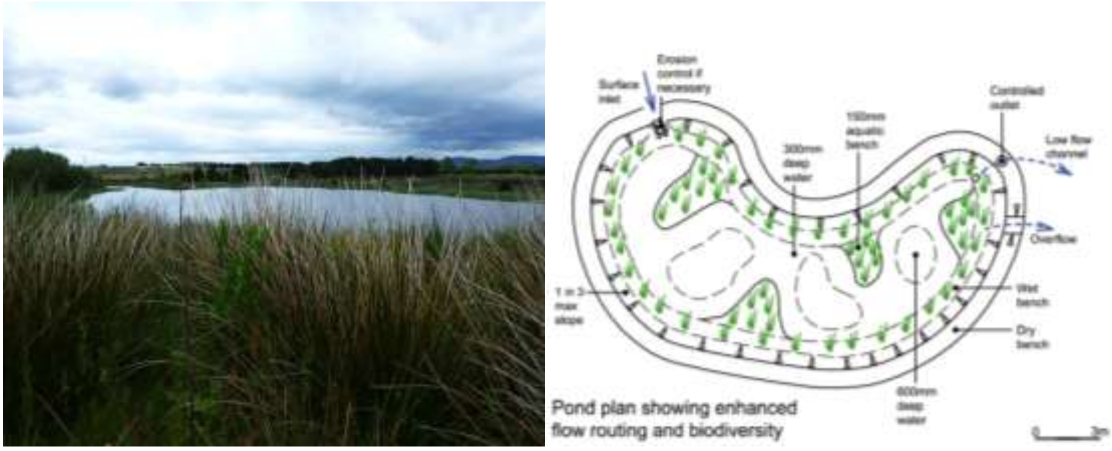
[1] SuDS Manual (CIRIA 2015)

[2] Ontario Ministry of Environment(), LID stormwater management planning and design guide for CVC (Dhalla and Zimmer 2010), LID technical guidance manual for Puget Sound (Hinman 2012), US EPA Stormwater best management practice design guide (Clar et al. 2004)

[3] Best Practice Environmental Management Guidelines (CSIRO 2006), Melbourne Water Constructed Wetland design guidelines (Melbourne Water 2015), DERM 2010.

2.6.4 Pond

SuDS ponds are temporary storage facilities that are continuously (perennially) wet. They are vegetated and used as both regional and end-of-pipe stormwater management measures (Woods-Ballard 2015). The aim of a SuDS pond is to attenuate the flow (ideally to peak discharge equal to pre-urban development levels) and delay the increased runoff volume created by upstream urban development. Implicit to the flow detention capability and the rapid reduction in flow velocity from high runoff rates (and high entrained sediment load) to low-velocity storage conditions, is that large particles will settle out of the water column to deposit on the pond bed. Figure 2.9 illustrates SuDS pond design (photograph: J4M8 SuDS pond).



(a) (b)

Figure 2.9 SuDS pond illustration. Figure 2.8(a) illustrates the established pond located within the J4M8 distribution park, Scotland, located at the downstream extent of the SuDS networks. Figure 2.8(b) is a schematic sketch of an indicative SuDS pond, modified from the from WSUD technical design guidelines (2006).

The prevalence of SuDS ponds across the UK, typically located at the downstream end of SuDS treatment trains means that they have received the most attention in previous research, focused upon examining water quality of the pond and its outlet discharge into the downstream receiving watercourse (e.g. Heal et al. 2006). Published data for this asset are summarised in Table 2.7, using 9 papers with large datasets; guidelines for expected pond treatment efficiencies are also provided.

Table 2.7 Published pond water quality and pollution removal levels (in %). Reported data are sourced from 9 international data sets; data ranges suggest multiple studies or repeat monitoring data are available, whilst single datum are from published an individual field samples.

Pollutant	Removal rate %	Reported or expected removal efficiency rate %		
		UK[1]	USA[2]	AUS/NZ[3]
TSS	20-99	67-81	60-90	85
Zn	5-74	84-89	44-60	-
Cu	4-99	67-82	26-60	-
Total N	35-60	31	30	45
Total P	76-82	48	20-73	65
Pb	10-85	-	54	-
K	-	-	-	-
Ni	53-78	57-87	-	-
Ca	7-37	-	-	-
Mg	24	-	-	-
Na	07-75	-	-	-
Cd	40-85	67-87	-	-
Cr	42-99	-	-	-
Fe	-	-	-	-
Ba	-	-	-	-
Sn	-	-	-	-
Mn	-	-	-	-
Al	-	-	-	-

References: Revitt et al 2004, Istenic 2012, Rushton 2001, Clary et al. 2011, Heal et a. 2000, Hossain et al. 2005, Reddy and Reddy 1993, Barret 2008, Istenic et al. 2012

[1] SuDS Manual (CIRIA 2015), Environmental Management Policy of Greater Dublin (Dublin City Council 2001)

[2] Ontario Ministry of Environment(1993), LID stormwater management planning and design guide for CVC (Dhalla and Zimmer 2010),LID technical guidance manual for Puget Sound (Hinman 2012), US EPA Stormwater best management practice design guide (Clar et al. 2004)

[3] Best Practice Environmental Management Guidelines (CSIRO 2006), Melbourne Water WSUD engineering procedures (Melbourne Water 2005), DERM 2010

2.6.5 Summary

The tabulated guideline and published SuDS sediment and pollutant detention efficiencies show a consistent disparity for all four assets under review. There is some disparity between international reported or expected efficiencies, disparity between literature reported efficiencies and the expected (or guideline reported) efficiencies of these SuDS and disparity between studies. The reason for these disparities may be e.g. environment (location, meteorological influence), asset design, variability in monitoring methodology; lack of knowledge of the complexity of internal SuDS processes themselves etc. The range of reported SuDS water quality treatment efficiencies is consistently larger than the expected efficiencies. The cause of the range in efficiency values is not definitively addressed within the literature; instead each study reports data within specific tightly controlled contexts (e.g. laboratory conditions, single event flows, single element (e.g. Pb) remediation, specific maturity age of asset etc.) which, typically, precludes both generic regional/national application of data and holistic scientific understanding of the black-box processes within the asset or full SuDS network. Thus, the context and constraints within which past studies were undertaken must be understood in more individual detail than that evident in the summary Tables; this is, therefore, the focus on the following review Section.

2.6.6 SuDS performance case studies

SuDS asset performance has undergone significant hydraulic analysis over the past 20 years, with one of the earliest studies being Yousef et al. (1985). Details of published SuDS case studies are provided in Table 2.8; this identifies the assets studied, temporal period of monitoring, and analytes for each study. Additional information on the field/laboratory testing methodologies, locations and additional salient information is provided in the subsequent text.

Table 2.8 Summary of SuDS field monitoring research The 42 studies summarised within Table 2.8 publish SuDS analysis of water quality improvement provided by specified SuDS assets. Studies occur in controlled laboratory environments and in the field, over a variety of durations and for a variety of pollutants. It is noted that the majority of water quality research focuses on rainfall-runoff event (defined as ‘event’ hereafter) analysis; considering the event mean concentration (EMC) or pollutant removal efficiency relative to flow or discharge. Studies identified by grey shading undertook sediment analysis (i.e. deposition or pollutant concentration monitoring or analysis). The International BMP database in included in the table for completeness, and is a key resource from which many of these key studies have been accessed.

Date	**Assets under review**	**Monitoring period**	**Analytes considered**	**Reference**
1985	swale, pond, detention basin	single event	8 months of runoff sampling. Analysis N, P, heavy metals (dissolved)	Yousef et al. 1985
1994	retention pond	long term non-event specific	sediment accumulation in ponds in the USA. Testing of pond sediment for TKN, N and P concentrations. Accrual is lower than guideline expected rates. Accrual is not linked specifically to rainfall, flow events or sediment loading	Yousef et al. 1994
1994	wetland	long term non-event specific	monitoring of sediment and removal efficiencies for summer and winter seasons	Kadlec and Hey 1994
1996	sand filters, swales, infiltration, bioretention, pond, wetland	single event	TSS, OC, TKN, TP, BOD, Zn, Cu, PAH, faecal coliform, Chlorides, some consideration of sediment deposition in swales	Claytor and Schueler 1996
1997	pond	single dry weather sample	Cu, Pb, Zn of deposited sediment particle size and density	Marsalek and Marsalek 1997
1998	swales, vegetated filter strips	single event	storm load pollutant removal of TSS	Barrett et al. 1998
1998	wetland	single event long term non-event specific	treatment of acid mine drainage inflow and outflow analysis. sediment was analysed for iron	Mitsch et al. 1998

Date	Assets under review	Monitoring period	Analytes considered	Reference
2000-2003	swales, permeable paving	single event	event based monitoring of water quality improvement monitoring of hydrocarbons, Zn, Ni, Cr, Cu, Pb, Cd (in suspension/solution), TSS, BOB, TN, Chlorine, TP, Ammonia	MacDonald 2003
2001	grass swale	single event	TSS, COD, TN, TP from stormwater runoff	Yu et al. 2001
2001	pond, detention basin, wetland swale, infiltration	single event	monitoring of inflow and outflow concentrations of sediment (TSS), Cu, P	Strecker et al. 2001
2001	wetland	single event	discharge monitoring of suspended and solute nutrient pollutants, COD, OP, N. analysis of monthly average removal rates	Jing et al. 2001
2002	wetland	multiple event	TSS, PSD relative to hydraulic loading,	Li et al. 2007
2002	wetland	single event	suspended and solute ammonia-nitrogen removal analysis	Thullen et al. 2002
2002	wetland	multiple event	150days of inflow chamber and outlet concentrations, plant roof uptake and soil of Al, Mn, Cd, Cu, Pb, Zn	Cheng et al. 2002
2004	ponds, detention basin, filter drains, swales, permeable paving, wetland	single event	event analysis of TSS, BOD, COD, N and P. sediment accrual in ponds (based on Heal et al. 2004, 2006 and Roesner et al. 2001 research) but not considered as a proportion of inflow or conveyance context. Sediment testing for contamination by Ni, Cr. No, CU, Zn analysis of plant uptake (accumulation in vegetation). weekly analysis rather than event analysis of selected pond discharge	Jefferies 2004
2004-2006	ponds, wetland	long term non-event specific	monitored the mass sediment deposition within the pond/wetland monitored sediment contamination levels of deposited sediment in the pond/wetland Monitoring for: TSS, TP, TN, total organic carbon, pH, ammonia-nitrogen, BOD, COD, Cr Cu, Zn, Pb, hydrocarbons	Heal et al. 2006, 2000
2005	grass swale and filter strips	multiple event	sediment deposition and resuspension monitoring across a grassed flow path. Multiple event analysis of sediment movement	Deletic 2005
2006	bioretention	single event	event pollutant analysis of Pb, Cu, Zn	Davis et al. 2006
2006	grass swale and filter strips	single event multiple event	monitoring of TSS, TP, TN removal during artificial event analysis	Deletic and Fletcher 2006

Date	Assets under review	Monitoring period	Analytes considered	Reference
2006	bioretention systems	single event	inflow and outflow for runoff events were tested for TN and TP	Passeport et al. 2009
			EMC analysis. EMC monitoring illustrates more effective reductions in warmer months than cooler periods.	
2007	bioretention	single event	stormwater inflow/outflow of TSS, TP, Cu, Pb, Zn, TN	Allen et al. 2007
2007	biofiltration	multiple event	monitoring of TSS, TP, TN removal by plant and infiltration in column experiments	Fletcher et al. 2007
2008	pond, detention basin	single event	TSS, COD monitoring of discharge and influent	Middleton and Barrett 2008
2008	biofiltration	multiple event	effluent analysis for TSS, TP, TN, Cu and Zn	Hatt et al. 2008
2008	biofiltration	single event multiple event	analysis of wetting and drying, sediment load and flow volume on pollutant treatment efficiency, infiltration analysis	Hatt et al. 2008
2008-2011	water butts, permeable pavement, green roof, swales, filter strips, under-drained swales detention basins, retention pond	long term non-event specific	quarterly samples taken of stormwater flow	Royal Haskoning 2012
		single event	analysis is focused on suspended and solute conditions. No bed deposition samples were collected. decrease in event based TSS through the SuDS treatment train (general reduction downstream).	
			General improvement in pollutant levels from the field site with SuDS implementation compared to the comparable untreated (Control) stormwater site	
2009	biofilters, bioretention	multiple event	TN, TP, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Zn, pathogens, event flow analysis (EMC and treatment achievement)	Bratieres et al. 2009
2009	biofiltration	multiple event	multiple event monitoring (wetting and drying schemes) on metal removal and metal concertation in filter media	Blecken et al. 2009
2009	biofiltration	single event multiple event	TSS, Cu, Pb, Zn monitoring of stormwater discharge	Hatt et al. 2009
2009-2010	bioretention basins, swales, wetland	single event	rainfall event based sampling of TSS, N, P, Al, CU, Pb, Zn	Parker 2010
2009	wetland, swale	single event	TSS, TN, TP analysis of discharge	Farrelly and Davis 2009

Date	Assets under review	Monitoring period	Analytes considered	Reference
2009	wetland, swale, bioretention, pond	single event	TSS, TN, TP analysis of discharge	Farrelly and Davis 2009
2010	bioretention, swales	single event	Portland green streets program, annual monitoring of artificial event discharge for TSS nutrients and heavy metals	Saltzman and Marriot 2010
2010	biofiltration	multiple event	heavy metals	Blecken et al. 2010
2010	pond, sedimentation tank	non-event	heavy metals (Cd, Cr, Ni, Pb, Zn), sediment and pollutants in suspension and solute. 3 defined monitoring occasions, non-event specific, within 1 year	Karlsson et al. 2010
2011	bioretention and grass filter strips	single event	influent and effluent monitoring of E.coli, enterococci	Hathaway et al. 2001
2011	detention basins, ponds	long term non-event specific	analysis of sediment pollutant concentration in deposited sediment within sediment forebays, monitoring of Cd, Cr, Cu, Fe, Pb, Ni, Zn. 17 of the 30 sites show exceedance of sediment guidelines for aquatic health and potentially toxic to the ecosystems	McNett and Hunt 2011
2012	bioretention, green roofs, permeable pavement,	single event	analysis of TSS, N, P, Cu. Pb, Zn in discharge. Some consideration of groundwater contamination is also provided	Dietz 2007
2012	bioretention	single event	23 events monitored for TKN, TP, TSS, BOD, Cu, Zn, Fe and Pb	Hunt et al. 2012
2012	swale, wetland, permeable pavement, filter strips	single event	monitoring of suspended sediment bound pollutants in effluent	Winston et al. 2012
2012	swale	single event	monitoring TSS, Pb, Cu, Zn, Cd, Cl. Stormwater runoff and swale discharge monitoring	Stagge et al. 2012
2013	bioretention	single event	event mean concentration analysis and load reduction assessment for P, N. An effort to consider multiple event export is made, with recommendations for monitoring of consecutive events	Brown et al. 2013
2013	swale	single event	rainfall event sampling of N, P, TSS, Cd, Cu and Zn	Knight et al. 2013
2014	wetland	long term non-event specific	intermittent monitoring over 6 years. Review of siltation (deposition)depth; TN, TP during EMC event analysis,	Merriman and Hunt 2014

Date	Assets under review	Monitoring period	Analytes considered	Reference
2014	rain garden, bioretention basin	non-event, single sample	heavy metal occurrence in infiltration zone, Pb, Cu. Non-event soil sampling of rain garden infiltration zone soil.	Quinn and Dussaillant 2014
2016	infiltration basin, swale	non-event, single sample period	sample of established SuDS assets on one date. Analysis of spatial distribution and concentration of Zn, Cu, Pb.	Tedoldi et al., 2016
ongoing	Bioretention, grass filter strip, swale, pond, wetland, infiltration basin, porous/permeable paving	Event, non-event, long term	USA based flow, solute, suspended and design database. Includes heavy metals analysis as well as e.coli. Majority of studies focus on solute or TSS results rather than deposited sediment pollution (2 settleable solid studies reported)	International BMP database; Clary et al. 2011; Barrett 2008; Whelton et al. 2016; Bustamante 2016

From Table 2.8 it is crucial to point out that there are few case studies that consider the deposition of sediment (and associated contamination of deposits); those identified relating to perennially wet assets (ponds/wetlands) are Heal (2000), Heal et al. (2006), McNett and Hunt (2011), Merriman and Hunt, (2014), Yousef et al. (1994), Marsalek and Marsalek (1997), Karlsson et al 2010; Tedoldi et al. (2016) assessed the deposited sediment pollution within infiltration assets (basin and swale); Deletic (2005) considers an ephemeral (filter strip and swale) system for event based sediment transport. Heal et al.'s (2004, 2006) studies are the closest in geographical similarity to those considered in the present thesis. Their 5 year study assessed sediment deposition in four prominent SuDS assets in the Dunfermline Expansion site (DEX) in Scotland (UK), to quantify annual accrual of sediment and changes to the associated contamination levels (specifically: Cd, Cr, Cu, Fe, Ni, Zn and Pb) (Heal et al. 2006). Only two assets illustrated accrual (5-7% volume water storage loss); one asset showed erosion/conveyance loss from the asset; one assets showed net equilibrium in the sediment storage balance. Of those with storage losses, these values are significant if upscaled to the asset design life ~25 years (i.e. losses up to 35% of the storage volume). Fe, Ni and Cr were typically within ±15% of the aquatic sediment and contaminated land guideline thresholds (Heal et al. 2006), demonstrating environmental hazard and risk in the waterbody and for maintenance/removal activity; other pollutants were within acceptable ranges.

In a similar monitoring campaign, McNett and Hunt's (2011) analysed sediment deposits in 30 forebay and outlet locations of ponds and detention basins in the USA. Of these, 17 locations breached US Environmental Protection Agency aquatic health sediment guidelines (Cu, Ni, Zn); yet, none breached land application of biosolid guidance (i.e. maintenance via sediment removal from ponds was not considered hazardous). Thus, despite only a handful of studies being completed there is clear evidence of sediment conveyance and detention within SuDS networks and a strong link to sediment carrier potential for high levels of metal pollution. However, the sampling regimes undertaken in these studies were single sample (Yousef et al. 1994, McNett and Hunt 2011), event based (Deletic 2005), bi-annually (Merriman and Hunt, 2014) or annual (Heal 2000, Heal et al. 2006); hence, no study reviewed how sediment transport (or detention timeframes) related to the rainfall events and source provision drivers of the system over the longer-term. As such, there appears scope to address this deficiency in a manner appropriate to empirically describing the longer-term trends and natural fluctuations in detention efficiencies appropriate to better informing remediation potential (timeframes) and SuDS maintenance strategies (accrual).

In terms of additional sediment transport investigation into SuDS, research has also considered sediment transport and water quality (TSS, TN and TP) within ephemerally wet filter strip and swale experiments, in Aberdeen (field and laboratory) and Brisbane (field) (Deletic 2005, Deletic and Fletcher 2006). These studies use the same empirical modelling tool (TRAVA, a 1-D overland flow and pollutant model) to analyse and simulate water quality and sediment detention. The sediment transport experiments considered the detention (during an initial artificial flow event) and re-suspension/re-deposition (during a subsequent artificial flow event) if fine sediment in a grassed filter strip (Deletic 2005). This research provided the first, and at that time only, sediment remobilisation analysis for urban sediment in a SuDS asset. Subsequently, Deletic and Fletcher (2006) also considered nitrogen and phosphorus concentrations for event specific discharges; field experiments simulated each different prescribed discharge, returning the flow to zero/baseflow between events (swale = 6 events, filter strip = 9 events, each provided with their own supply load). Given the ephemeral nature of the assets, this can be considered the first simulation of multi-event flow sequence scenarios of flow, sediment and pollutant conveyance in SuDS assets, although with a noted focus on suspended sediment concentration at the in/outlets rather than deposition within the SuDS assets. The data showed that post-deposition flows generate minor sediment loads, indicating sediment detention to be temporary. Successive sediment laden flows resulted in different discharge pollutographs (i.e. different volumes and rate of sediment conveyance) due to changes in flow (and propensity to entrain) and sediment availability (and supply). Yet, TRAVA was found to effectively model the individual entrainment-transport-deposition events through the use of particle size and fall velocity within the empirical analysis. Despite this research being undertaken in a tightly controlled laboratory type setting (using known artificial flow and load provision) it does provide a platform for future, multiple event sediment transport and pollution research in other (non-overland flow) SuDS assets.

The published research to date provides evidence of event based water quality treatment by SuDS and insight into sediment deposition within wet assets. However the link between source-sink has not been completely unravelled, possibly due to the complexity of urban (fine) sediment transport monitoring or tracing. Published water quality research has primarily focused on either the influence of SuDS on event based water quality change or non-event analyse of in/effluent suspended and solute pollution concentrations. Sediment loading, movement (as the primary carrier of urban pollutants) has been analysed, but there is an opportunity for source-to-sink analysis of water quality in conjunction with sediment deposition/conveyance. This case studies review illustrates a research gap in linking all the stormwater remediation processes together (source-sink, multiple event movement and bed deposition) and the opportunity for a future research to consider SuDS sediment pollution in a holistic manner; solute, suspended and deposited pollutant changes due to SuDS influence.

2.7 Pollutant remediation processes

Whilst the objectives of this thesis are not specific to analysis of the biogeochemical of remediation potential within SuDS, it is prudent here to provide a summary of the processes by which correlations between sediment detention and pollutant concentration correlations may show spatial and temporal variation. SuDS are designed as passive water quality treatment assets (Woods-Ballard et al. 2015) using physical interventions and bio/phytoremediation methods; these been summarised in Appendix I specific to the urban pollutant focus of this thesis. Three remediation methods (adsorption, bioremediation and phytoremediation) occur without intervention, due to the design of SuDS assets and its control of flow/sediment/influent.

2.7.1 Adsorption

Adsorption is a common pollutant management method designed to remove soluble pollution from influent (such as stormwater) and fix it to particulates. This results in the contamination of particulate matter; i.e. sediment and organic material that is temporarily detained within the remediation asset (Peng et al. 2009). Crucial to asset management, these sediment adsorbed pollutants remain within the SuDS assets for longer time periods than solute (due to flow conveyance; Section 2.4); whilst this may provide better potential for remediation, if the rate of remediation is less than the accrual rate the level of toxicity of the sediments will rise (see Section 2.6.6).

Na, Zn, Cd, Cu are generally highly available in stormwater solution. Pb, Fe and Al bind strongly to sediment and are not easily available in solute form (Sansalone and

Buchberger 1997). Cr, Fe, Cd, Ni, Zn, Cu, Mn, Mg and Ba have also been reported to effectively become adsorbed from polluted influence onto sediment particulates (Lee et al 2011, Niu and Bolesky 2001, Bailey et al. 1999, Peng et al. 2009, Garnier et al. 2006). Sn in elemental form is general not easily soluble in water, but chloride (II) forms are soluble and organotin is noted to become adsorbed to sediment (Fent 1995). In addition, the use of chelating agents or surfactants assists with adsorption success. This addition of minerals such as apatite or zeolites alters the immobilisation of the metal pollutants; e.g. apatite is reported to effectively fix Pb, Mn, Cu, Cd, Zn, Mg, Ba, Na (e.g. Peng et al. 2009), whilst Ca and P may be released.

Time has been identified as a key influence in adsorption activities. The majority of adsorption occurs during the first 48 hours of the pollutant influent introduction to fine sediment. However, this fast desolubilisation is not found for all urban pollutants; e.g. up to 30% of Cd adsorption requires a longer time period, up to 30 days. Furthermore, the adsorption bond between sediment and Mn in natural river settings has be recorded to decline in strength after 30 days, resulting in a small and slow release of adsorbed Mn pollutant into solution or re-adsorption onto a new sediment particle (movement) (Garnier et al. 2006). Thus, knowing the time of sediment detention within a SuDS asset is clearly crucial to estimating and modelling pollutant concentration change due to remediation.

2.7.2 Bioremediation

Bioremediation is the use of organisms to breakdown, transport and/or remove pollutant contamination (Romantschuk et al. 2000). Bacterial activity is the primary method of pollutant degradation while algae, fungi, micro-organisms including small insects and worms and plants form part of the bioremediation participants. Bacteria and fungi degradation occurs as change in speciation and decomposition of pollutants to productive soil material due to bacterial and fungi activity in the soil. The bacterial and fungal activity can, with stimulation from plant exudate (sugars, carbon, amino acids, carbohydrates etc.), function as chemical attractants for pollutants and alter their toxicant adsorption, bioavailability and leachability.

Bioremediation has been found to be effective for the control and/or removal of range of urban pollutants, specifically: Fe, Cu, Zn, Mn, Al and Cr (Rascio and Navari-Isso

2001). Reported pollutant availability for bioremediation (i.e. the ease that organisms can access pollutants) follows the trend: Pb <Cd <Cu <Zn <Mn <Ni <Cr (Rascio and Navari-Izzo 2001), due to variability in their speciation, mobility and activity. Bioavailability is very sensitive to local environmental conditions, such as pH, humic acid, temperature etc. (Section 2.5.4), and it is worth highlighting that bioremediation of metals is more effective in anoxic conditions of asset sludge layers (Section 2.2) where bacteria dissolve metals without creating acidic conditions (Mulligan et al. 2001). This final point is important, as SuDS assets with deeper bodies of stagnant water (e.g. ponds, wetland) are more likely to exhibit anoxic conditions in the bed sediment layers.

2.7.3 Phytoremediation

Phytoremediation, the use of plants to control, treat and/or clean polluted influent and soils, is a form of bioremediation. It is a multi-stage process including: stabilisation of the pollutant; extraction from the water/fluid/soil/sediment; speciation change of the pollutant. The change of a pollutant structure (e.g. a change or flux of N between: nitrate (NO_3^-), nitrite (NO_2^-)), known as speciation change, can occur as a result of plant interaction with the pollutant. This is further influenced by related processed of the rhizosphere, translocation and chelation potential (Pilon-Smits 2005). In short, phytoremediation can be achieved by *direct uptake* or *speciation change* of the pollutant. As direct uptake requires that the pollutant be a solute or easily soluble, desorption processes to release the pollutant from the sediment would have to occur first (e.g. via a catalyst (such as high water acidity, preferential sorption to an alternative available element), microbial/bacterial or exudate (i.e. plant or root excretion) activity (Yuanwen et al. 2003)).

Stormwater pollutants known to illustrate phytoremediation benefit are: N, P, Cl, Cr, Cd, Zn, Pb, Ni, Cu (Suthersan 1997). Of these P, N, Cu, Zn and Ni are elements essential for plant growth and the two former metals are generally highly available in stormwater solution for easy uptake; Cu has, therefore, been found to have a higher phytoaccumulation rate in plants than other metals (Wu et a. 2014). Conversely, Cd, Cu, Pb and Cr have no reported essential biological function and Pb binds so strongly to sediment that it is not readily available in solute form (Sansalone and Buchberger 1997). Previous studies have found that Cu and Pb (and P) phytoaccumulation is better promoted when the roots of phytoremediation plants are waterlogged (saturated) for very long periods of time; while Cd recorded the lowest phytoaccumulation rates of the metals tested (Wu et at. 2014).

Whilst this evidence helps provide insight to explain long term trends in SuDS sediment pollution, it also explains the preference of SuDS planting regimes towards hyperaccumulator plant species (Prasad and Freitas 2003, Juwarkar and Yafav 2010). An ecological review of phytoremediation is not the intention of this thesis, however it is prudent to demonstrate awareness of the pollutant-specific remediation potential of this process to better explain pollutant changes over the long-term monitoring timeframe considered herein. Specific species are often found to accumulate one specific pollutant effectively, for example the Brassicaceae genera (Brassica) has 72 species that hyperaccumulate Ni and 20 species that hyperaccumulate Zn. However, there are certain species that hyperaccumulate multiple pollutants. *T.caerulescence* accumulates Cd, Ni, Pb and Zn. *T. goesingense* and *T. ochroleucum* accumulate both Ni and Zn and *T. rotundifolium* accumulates Ni, Pb and Zn (Prasad and Freitas 2003). 75% of hyperaccumulators accumulate Ni but there are very few (5 reported species) that can accumulate excessive Cd (Juwarkar and Yafav 2010). Current understanding suggests hyperaccumulation efficiencies follow Zn, Mn>Ni, Pb>Cd (Juwarkar and Yafav 2010, Rascio and Navari-Izzo 2001), providing important insight into the relationship between SuDS treatment efficiency and asset-specific planting regimes.

2.7.4 Influential environmental factors on adsorption, bio and phytoremediation

Adsorption, desorption, bioremediation and phytoremediation are all sensitive to environmental conditions. Acidity, saturation and temperature influence the activity and efficiency of these remediation methods (Peng et al. 2009). Cd and Zn are preferentially released from adsorbed conditions with acidification of the environment compared to Cu and Pb (Calmano and Forstner 1993). Metal pollutants are generally more easily mobilised (released from the bond of adsorption to sediment) when the environment is oxidised (Calmano and Forstner 1993). Redox reactions (and therefore speciation) are also a controlling factor in metal pollution mobility and is interlinked with acidity and oxidation influence on adsorption retention of pollutants by sediment (Calmano and Forstner 1993). Numerous other metal pollutants adsorption to sediment are noted to have pH limitations. Fe and Al are noted to be most pH tolerant (pH 2.5) while Cd and Zn have low (pH 6) tolerance of acidification (tolerance: Fe, Al>Pb>Cu>As, Ni>Cd, Zn) (Peng et al. 2009). In addition, temperature can influence adsorption of metals to sediment and thus the availability of pollutant for bio/phytoremediation. An increase in temperature has been found to create a slow decrease in pollutant adsorption to sediment (Peng et al. 2009) suggesting a seasonal influence (e.g. more effective adsorption during winter months). A general awareness of all these process controls is important when considering annual monitoring and analysis across geographically distinct SuDS case studies, as within the present thesis.

2.8 A review of current pollutant movement monitoring methods

A range of sediment monitoring methodologies exist. Table 2.9 provides an overview of established sediment tracing methods, most commonly used for agricultural point/diffuse pollution study and river bank/bed morphodynamic research. Sediment tagging methodologies extend from basic marking procedures, such as painting natural sediment (Ingle 1966, Hassan and Ergenzinger 2003), through to more high-tech forms such as artificial particles with magnetic or coloured cores for in-situ monitoring (Black et al. 2007). The persistence of tracers in the environment varies significantly according to composition and therefore is it necessary to ensure the selected tracer is appropriate to the research objective requirements.

To accurately and effectively monitor urban sediment transportation characteristics several specific tracer requirements are recommended to: (i) have negligible detrimental impact on the environment where they are released and in all influenced areas downstream; (ii) remain active in the field for an extended period of time (months to years), sufficiently long enough to examine transport during multiple events; (iii) mimic the characteristics and transport dynamics of naturally occurring urban sediment; (iv) have release locations appropriate to the source characteristics (impervious/ pervious/vegetation) and pathways to be monitored; (v) independently identify different point sources (e.g. roof, road, commercial, residential – Section 2.8.4) of sediment/pollutant.

Thus, tracing methods fulfilling these requirements (even without demonstrated urban application) have been further reviewed and summarised in Table 2.9. This stems from review of a body of 28 papers available in the technical literature on this subject; for each, the Table provides opinion on the viability of tracer use for the SuDS application required to meet the objectives of the present thesis.

Table 2.9 Overview of established sediment tracing methods. Multiple (28) publications have been reviewed and summarised within Table 2.9. References specific to Table 2.9 are provided in the footnotes[9].

Trace Method	Number of Identifiers	Activity Period in Natural Environment	Recorded Use	Potential for Utilization in Urban Environment
Radionuclides	numerous	30–40 years	Study of erosion and deposition in the landscape, chronometer for sediment deposition in ponds, lakes and floodplains, agricultural sediment erosion, catchment erosion and deposition in lakes.	Effective. Long activity time results in potential difficulty in replicability. High resource requirement.
Fingerprinting	numerous	Natural particle life cycle	Watershed/ catchment scale sediment budget analysis. Sediment source analysis.	Effective but requires chemical signatures to be significantly different between sediment sources. Requires technical support and laboratory equipment (AAS) and sampling for numerous chemical concentrations.
Painted/coated natural particles	numerous	Limited time frame due to low trace adhesion/adsorption to sediment particle. Solar degradation may shorten field activity period.	River bank erosion, sediment transport though fluvial networks, larger sediment, pebble and gravel tracing.	Highly visible. Difficulty in separating coated material from remaining sample sediment.
Magnetic particles	1	Extended dependent on synthetic material (coating) chosen or natural magnetism	Soil erosion within a watershed. Sediment loss and detachment from source.	Artificial material limiting natural assimilation or breakdown. Natural magnetism has limited unique signatures.

[9] Black et al. 2007, Caitcheon 1998, Carter et al. 2003, Clark 1986, Collins et al, 2010, Davis and Fox 2009, Deasy and Quinton 2010, Ferguson et al. 1996, Gymer et al. 2010, Hassan et al. 2003, Ingle 1966, Ju et al. 2013, Kayhanian et al. 2012, Li et al., 2006, Mabit et al 2011, Mahlet et al. 1998, Matisoff et al, 2002, Napier et al. 2007, Parsons and Foster 2011, Poleto et al. 2009, Sloan and Gries 2009, Timperley et al. 2005, Ventura et al. 2001, Walden et al. 1997, Zapata 2003, Zhang et al. 2009, Zhang et al. 2001, Zhang et al. 2003, Zhu et al. 2010.

Trace Method	**Number of Identifiers**	**Activity Period in Natural Environment**	**Recorded Use**	**Potential for Utilization in Urban Environment**
Magnetic fluorescent material	4	Extended dependent on the particle material. Fluorescent activity is extended due to the particle coating and design	River sediment transport. Piped network sediment transport.	Supports monitoring without loss of material from the field environment. Easily separated from total sample sediment. Highly visible.
REO	17 (15 readily analysed)	Extended (months–years)	Particle translocation. Surface erosion, due to rainfall-runoff, overland flow, sediment transport from multiple sources. Agricultural erosion. Solute/suspended sediment redistribution in snow, ice, urban, agricultural and rural environments.	Not visible. Limited environmental impact. Significant identifiers. Shown to be effective in alternative conditions. Untested in the urban environment but meets urban monitoring requirements.
Pollen	Limited to natural vegetation pollen availability	Annual time frame (not event specific) to decades	Vegetation and land use histories (chronometer). Pollen peak correlation with annual sediment erosion and deposition. Ability to trace sediment to source, when source is from natural (vegetated) surfaces.	Limited due to activity period limitations. Complexities relating to urban surface type, urban source and grassed/vegetated areas that comprise the SuDS.
Synthetic/ artificial particulates	limited	Extended (similar to natural particles)	Mass transport flux, TSS concentration and bed load change.	Difficult to consider source to sink movement unless limited to a single source within the network under consideration, due to limited identifiers. Replicability difficulty may not effectively mimic natural sediment characteristics.

Trace Method	Number of Identifiers	Activity Period in Natural Environment	Recorded Use	Potential for Utilization in Urban Environment
Total Suspended Solid balancing and PSD analysis	limited	Extended (similar to natural particles)	Mass transport flux, suspended solid concentration change, PSD change related to influence of rainfall and source contribution (high level).	Limited to flux and balance analysis. Difficult to identify source from PSD and mass change alone.

2.8.1 Fingerprinting

Chemical fingerprint analysis of soil to identify sediment source has been extensively used in large scale sediment balance analysis. A chemical signature for each source is defined using multiple element concentrations. Poleto et al (2009) used 11 variables to create four sediment source fingerprints (As, Zn, Co, Pb, Cd, Ni, Mn, Cr, Cu, Fe and TOC). Non-parametric analysis, the Mann–Whitney U-test, Kruskal-Wallis H-test and P-value analysis, are commonly completed to identify effective fingerprint elements and to define results (Poleto et al 2009, Carter et al. 2003). All sediment sources contributing to the location of interest and identification of effective unique signatures (chemical elements) for each sediment source (Davis and Fox 2009) must be acknowledged in sampling design and analysis. Thus, sediment fingerprinting is highly effective in identifying catchment level sediment contributions to a river system, where source differentiation is significant (e.g. urban vs agricultural vs forested land use) (Walling 2005) e.g. a concrete batch plant may provide a site specific signature including significant Ca and Si spikes within an urban signature. However, where source differentiation is more difficult (e.g. intra-urban area or where analysis requires separation of main road versus car park sources), a greater number of chemical elements may be required to define the unique source fingerprint or signature (Collins et al. 2010). Fingerprinting has not been used in source-to-sink tracing of sediment movement in SuDS, possibly because of this complication in identification between different urban surfaces. In conclusion, urban sediment source to sink tracing using fingerprinting is difficult for detailed source-sink SuDS analysis, nor can it be applied where discrete time-stamped tracing of repeat releases from a single source is required.

2.8.2 Fluorescent (magnetic) artificial or coated sediment

Commercial tracers, developed over the last decade, provide specified particle size, distribution and density material with magnetic and/or fluorescent identifying properties. This material has been successfully employed to mimic natural particle characteristics and transport in urban watercourses, drainage networks, rivers and coastal tracing studies (e.g. Black et al. 2007, Ferguson et al. 1996). Due to the fluorescent nature of this material in-situ non-destructive monitoring can occur using field fluorometers; this allows continuous monitoring of tagged sediment concentration without loss of material from the system due to sampling. Similar to the dye trace functionality of Rhodamine WT, fluorescent sediment can provide a plume and concentration curve for specified monitoring locations within the reach of interest (Guymer et al. 2010). The magnetic property (para-magnetism) enables active magnets to separate out tagged sediment from total sample or flow load (Sloan and Gries 2009) for ease of sampling and, potentially, retrieval. The fluorescence variation (four separate fluorescent signatures) supports non-destructive analysis using the excitation and emission wavelengths (pers. comm. J Pollyket, 2013). In conclusion, although para-magnetic fluorescent (PMF) tracers appear an effective visual sediment transport marker within traditional urban drainage pathways (culverts, pipes and permanently flowing drainage channels) there is no evidence in the literature of their use within ephemeral vegetated urban pathways such as those found within SuDS. The limitations and benefits of this material for SuDS application therefore remain unknown.

2.8.3 Radionuclide and radioactive isotope trace use

Radionuclides have been used to track sediment movement in two ways. Fallout radionuclides result from historic radioactive events (e.g. Reactor 4 Chernobyl, Russia 1986) and the concentration of these (^{137}Cs, ^{7}Be, ^{226}Ra, ^{222}Rn and ^{210}Pb) in soil samples can indicate the redistribution of sediment over the intervening time period (Guzman et al. 2013). Radioactive isotopes have also been used as introduced sediment tracers. Wooldridge (1965) sprayed ^{56}FE, a naturally occurring radionuclide, on to soil to study soil transport (Zhang et al 2001). ^{60}Co has also been used as a ‘manufactured’ soil tracer, added to soil in solution form (Toth and Alderfer 1960). The use of this type of radionuclide generally requires local authority approval prior to use due to environmental and health and safety concerns. Although used in the UK in the past, concerns regarding use and release of radioactive material (Parsons and Foster 2011) have reduced this methods popularity. Exception to this is gamma (γ) ray emitting radionuclides, which are commonly used isotopes to tag and trace sediment from source to deposition (Zapata 2003). These are a range of isotopes, providing a spectrum of individual identifiers, and sampling does not significantly disturb the monitored environment. That said, sampling and analysis equipment are noted to be expensive, requiring specific sample preparation and gamma spectrometry equipment. As a result, use of radionuclide tagging is not currently common practice in the natural, urban or watercourse environments of the UK.

2.8.4 Rare earth oxide trace use

Rare Earth Oxides (REOs) are elements naturally found within soil and bed material. They form the lanthanide group of elements (15 lanthanides, scandium and yttrium) within the periodic table and are classified as rare due to their very low concentration (parts per billion, ppb) within the shallow layers of the earth's crust. Such low concentrations require strong acid digestion and assessment by Inductively Coupled Plasma Mass Spectrometer (ICPMS) (Zhang et al. in 2001, Polinares 2012); this is possible only for 15 of the 17 REOs available. Thus, two methodologies have been developed and tested for REO use in the natural environment; firstly, natural material can be REO tagged in a laboratory environment and released on site manually in known locations, concentrations and particle distribution; secondly, REO solution can be sprayed directly onto field surfaces to adhere to sediment in-situ (Zhang et al. 2001, Deasy and Quinton 2010).

The use of rare earth material to tag and monitor sediment was first intensively studied and published by Zhang et al. (2001). Here five lanthanides were used to tag silt loam soil and exposed to leaching and concentration testing relative to particle size. The REOs were found to adsorb evenly to all tested particles sizes (>53µm - >2mm) and no significant movement of REO occurred from tagged to non-tagged material during the leaching process. As such, transport and erosion tests were undertaken (Zhang et al. 2003) to identify rainfall driven soil erosion down a 10% unvegetated slope. The concentrations of sediment from known positions up the slope were calculated from runoff samples, and a laser survey identified surface soil loss after rainfall events. In a similar experiment, Polyakov and Nearing (2004) also found REO to be effective in quantifying the soil redistribution on a slope and sediment loss/erosion due to rainfall-runoff events. These studies suggest that, although not tested in SuDS systems *per se*, REOs are a viable method by which to trace sediment movement on natural wetted surface or within flow systems.

REO material is naturally adsorbed effectively to fine sand - clay soil fractions (Tyler 2004). Adsorption can be influenced by pH (acidic conditions decrease adsorption), the presence of humic material and carbonate ions (high humic and carbonate iron conditions decrease adsorption) (Tang and Johannesson 2010, Johannesson and Hendry 2000, Tyler 2004). Rare earth elements are naturally found as phosphates, silicate, fluorides and carbonate minerals, in granite, pegmatite, gneisses, magmatic and metamorphic rock (Tyler 2004).

2.9 Research needs identified from the literature review

This literature review has identified that the source of urban pollution (sediment, metals, minerals and nutrients) is well documented with a general understanding of the washoff of diffuse urban pollution from urban surfaces into stormwater management assets. Although relatively small in number, there do exist field and laboratory studies on rainfall-runoff correlations to pollutant washoff, focussed at an event-scale of analysis. With regard to design guidelines and associated evidence base for SuDS, review of published SuDS efficiency studies highlights both, the large range of efficiencies identified from field data and, a disparity between the reported results and expected SuDS efficiencies in legislative guidelines (nationally and internationally).

Limited information is available on SuDS sediment pollution deposition accrual (with the exception of ponds) or the longer term contamination of this sediment. The influence of multiple events on sediment re-suspension and deposition has been only tentatively considered (Deletic 2005, Deletic and Fletcher 2006) for grass filter strips and swales, but over a very limited number of events with strong focus placed upon tightly-controlled laboratory-derived flume-based experimental study. The longer term influence of rainfall-runoff events, and the potential for long term retention of deposited sediment within SuDS assets has not yet been considered in detail. Thus, while SuDS are expected to have a functional life-cycle of 25-30 years, the efficiency of SuDS assets over this period appears unknown.

Current best practice in SuDS design uses first order kinetic decay analysis (in modelling tools such as MUSIC or SWMM) yet, review of this method identifies limited inclusion of the classical sediment transport characteristics fundamental to accurate prediction. Only a few studies evidence validated empirical modelling of multiple event sediment movement across grassed filter strips and swales (Deletic 2005; Deletic and Fletcher 2006); however, this work does not consider perennially wet SuDS assets (wetland, linear wetland, ponds).

Despite the majority of urban pollution movement occurring through fine sediment transport (heavy metals, minerals and nutrients adsorbed to fine sediment particulates) current research does not appear to have investigated the potential link between SuDS sediment transport and pollutant remediation.

When this literature-derived knowledge base is taken as a whole, it highlights that further research is needed to understand the longer term sediment pollution stormwater remediation efficiencies of SuDS, crucial to asset performance, maintenance and design life (e.g. decommissioning). Specifically there is need to:

- Identify the parameters that influence SuDS sediment and pollution detention and remediation efficiency to help explain the disparity in report and guideline efficiencies and inform future efficiency expectations;
- Examine whether single-event sediment pollution monitoring provides adequate evidence of SuDS efficiencies or if multiple-event monitoring is necessary to illustrate longer term SuDS efficiencies;
- Identify whether multiple rainfall-runoff events influence sediment pollution detention within SuDS assets and examine whether the post-deposition events result in continued sediment conveyance;
- If continued sediment pollution conveyance occurs, the quantification of sediment pollution movement (detention and discharge) longer term is required so as to inform maintenance requirements and future design improvements;
- Examine sediment contamination within SuDS assets (beyond existing knowledge restricted to ponds) to identify if detained urban pollutants are being remediated or concentrated within SuDS; and,
- Provide field based research that considers both water quality (discharge pollutant concentrations) and sediment deposition/release in a unified monitoring and analytical context.

3 Field monitoring and analysis methodology

3.1 Aim of data collection

The purpose of this methodology chapter is to introduce the approach taken to create, collect and analyse fine sediment pollution from within multiple established SuDS assets. The methodologies presented within this chapter have been selected and implemented across the chosen field sites to create a novel and detailed dataset examining the sediment pollutant movement, deposition and conveyance through SuDS. To examine multiple event sediment pollutant movement it is necessary to employ a method to trace sediment from source-to-sink. This provides a time/rainfall/flow referenced sediment movement dataset that can be dissected and examined to look to links, explanations and further understanding of the multiple event sediment pollution detention and conveyance of SuDS assets/networks.

To examine the detention capabilities and multiple rainfall-runoff event sediment movement a dataset illustrating sediment deposition and discharge is first required. Field sampling and sediment tracing methodologies have been outlined in this chapter to define how the field sediment detention dataset was created. The aim of the data collection was to provide (i) suspended and deposited sediment dataset illustrating the total sediment deposition and mass balance, (ii) time stamped, traced sediment movement and deposition dataset illustrating sediment movement from source to sink, and (iii) a sediment deposition dataset that could be analysed for sediment contamination. Field sampling has been undertaken, and data collected, to provide this novel SuDS sediment dataset.

A major part of the methodology development for this field research justifies and validates the development of an effective REO tracer method for fine sediment transport measurement within urban catchments and SuDS. The use of REO tracers has been combined with: suspended and bed deposition load analysis; particle size analysis and pollutant (heavy metal and mineral) concentration analysis helps identify sediment detention efficiency and associated contamination levels. Methodology on the measurement of the key sediment/flow/rainfall parameters that influence sediment movement within the catchment and SuDS are also described.

3.2 Rationale for field site selection

To provide context to the methodology development an overview of the field sites monitored as part of this thesis research are initially presented. An overview of the three field sites selected for this thesis' study is summarised in Table 3.1. Each site was selected for specific purposes but share similar field sampling and activity methodologies.

Table 3.1 Overview of research field sites and analysis

Field Site	**Rainfall-runoff events**	**SuDS assets monitored**	**Analysis undertaken**
Campus swale	3 artificial flow events	Swale	• 2 REO tracers released onto 1 urban surface
J4M8 distribution park	12 months of naturally occurring rainfall-runoff	Swale Wetland Linear wetland (Pond)	• 12 REO tracers released onto 3 urban surfaces • PSD • Mass TSS/bed deposition • Heavy metal and mineral sediment pollution concentration
NGP development	6 months of naturally occurring rainfall-runoff	Pond	• 8 REO tracers released onto 1 urban surface • PSD • Mass TSS/bed deposition • Heavy metal and mineral sediment pollution concentration

Campus Swale: Justification for the inclusion of the small-scale pilot study of Heriot-Watt's campus swale (Section 3.3.1) was specific to development, testing and validation of the REO tracer field methodology (Section 3.6), prior to its use in larger field sites with full SuDS network. An overview of the Campus field site is provided in Figure 3.1. The swale provided a single, ephemeral, established and maintained SuDS asset, where intense monitoring and tight-control of environmental factors were possible. Artificial flows of known hydrograph and sediment releases of known location, concentration characteristics and flow paths could be conducted without influence from land use activities or disturbance to local land users. Hence, the objective of these controlled pilot tests was to identify whether the REO tagging methodology: (1) provided measurable tracer concentrations in sediment samples representing sediment transport within this asset; and, (2) to select the tracer concentration appropriate for use.

J4M8 Distribution Park: Section 3.3.2 provides the full details of the site while Figure 3.2 provides an overview of the site, which was the primary location for SuDS network monitoring (Table 3.1) in the present thesis. This field site was selected for four specific reasons: (1) SuDS design follows current best practice for installation; (2) there are multiple SuDS networks on site, each with different asset treatment trains and contributing land-uses, permitting network comparison in equivalent environmental settings; (3) the SuDS are established and have been maintained for the past 10 years) (Berwick 2012, West Lothian Council 2013); (4) local land-use is urban, typically commercial, with variable source (roof, road, car park).

NGP Development: Located in Newcastle, England this additional field site was feasible via case study city agreements formed in 2014 via the EPSRC Blue-Green Cities consortium. An overview of the site is presented in Figure 3.3. This was considered beneficial in the present thesis, to provide additional information specific to constructed SuDS ponds. Although there is a pond located at the downstream extent of the J4M8 stormwater network, it is very large (16,240m^2 surface area), deep (over 1.5m) and affected by stormwater runoff from neighbouring urban catchments (other than those managed by the SuDS network discussed above); this presented a safety risk for monitoring access and added complexity of analysis. Thus, Newcastle Great Park (NGP; Section 3.3.3) provided an optimal alternative pond site, as this forms the only, isolated SuDS asset for the entire (residential/commercial) urban development area, i.e. there are no upstream SuDS network(s) to consider. The NGP pond was included in this research for the follow key reasons: (1) the development area included residential land use, thus providing field verification of land use sediment and pollutant contributions for residential areas (complementing the more commercial use of J4M8), (2) the development provided full and safe monitoring access across the 2,400m^2 surface area and depth generally less than 0.5m; (3) it is and established and well maintained asset, similar to J4M8.

Figure 3.1 Overview of the Campus Field Site

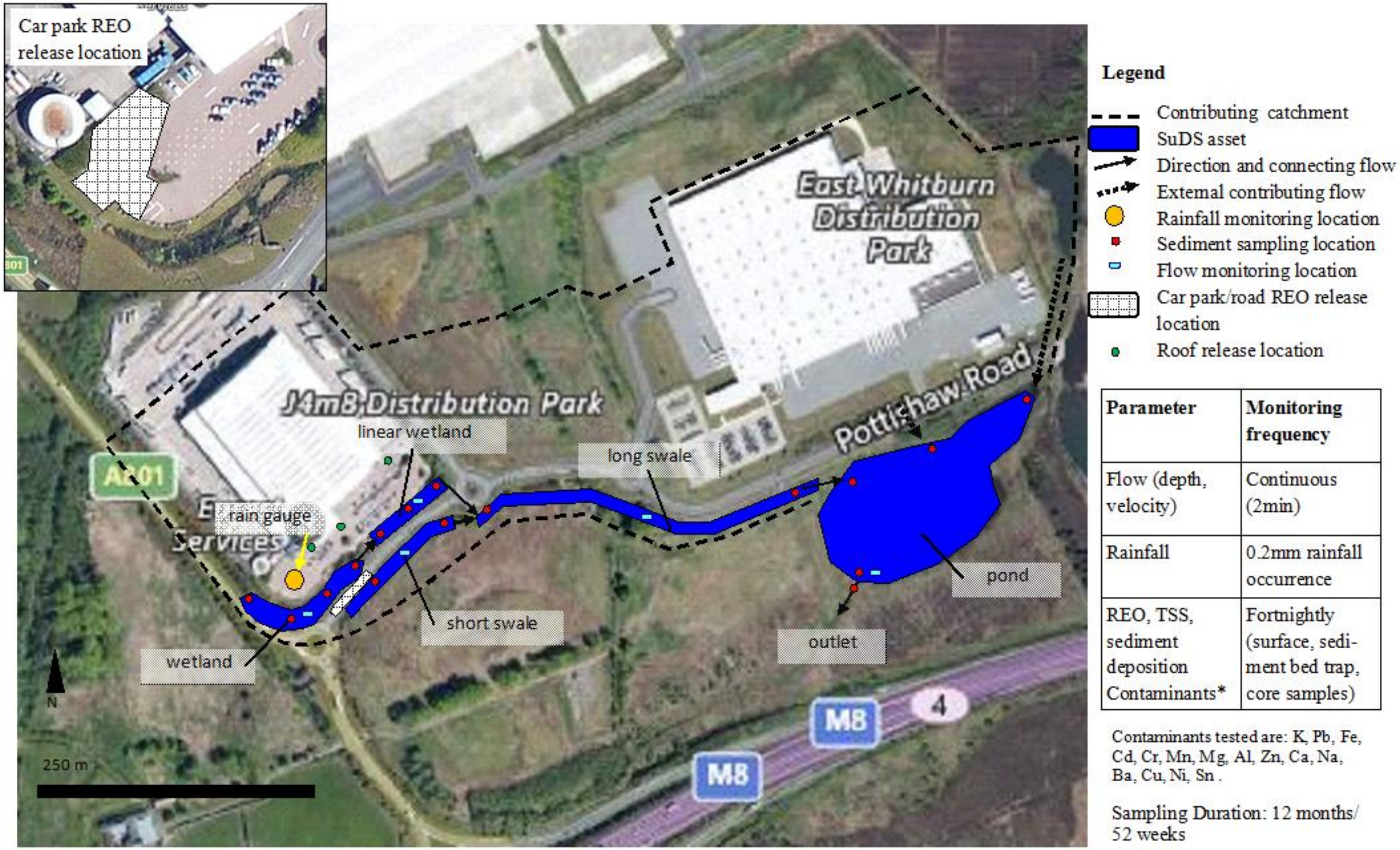
Legend

- Contributing catchment
- SuDS asset
- Direction of flow
- Rainfall monitoring location
- Sediment sampling location
- Flow monitoring location
- REO release location

Parameter	**Monitoring frequency**
Flow (depth, velocity)	Continuous (2min)
Rainfall	0.2mm rainfall occurrence
REO, TSS, sediment deposition	Event: 5 mins Inter event : 1 hour post flow Post event:: 1 week, 6 months, 12 months

Sampling Duration: event and 12 months

Figure 3.2 Overview of the J4M8 Field Site

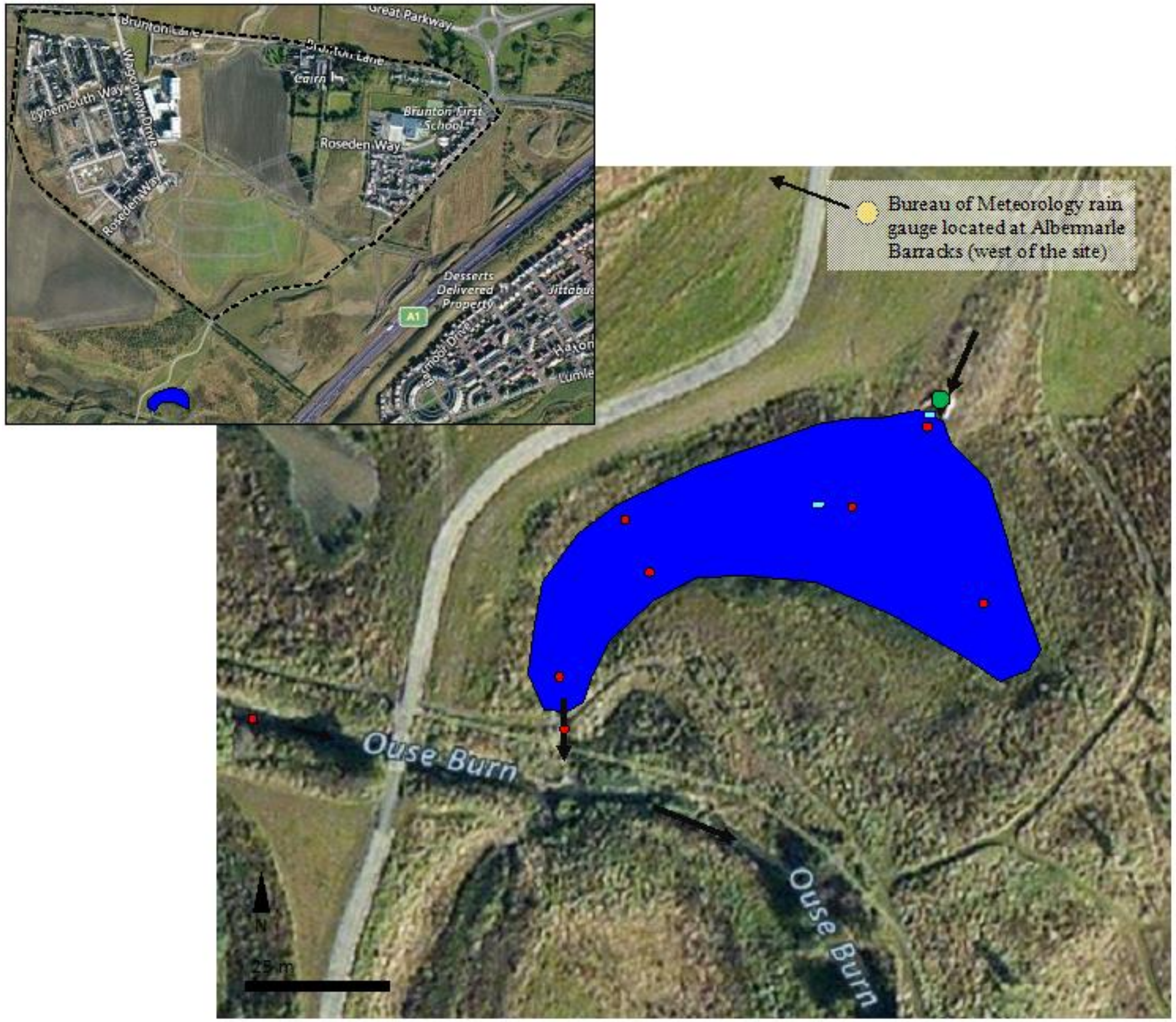


Parameter	**Monitoring frequency**
Flow (depth, velocity)	Continuous (2min)
Rainfall	0.2mm rainfall occurrence
REO, TSS, sediment deposition Contaminants*	Fortnightly (surface, sediment bed trap, core samples)

Contaminants tested are: K, Pb, Fe, Cd, Cr, Mn, Mg, Al, Zn, Ca, Na, Ba, Cu, Ni, Sn .

Sampling Duration: 12 months/ 52 weeks

Figure 3.3 Overview of the NGP Field Site



Parameter	**Monitoring frequency**
Flow (depth, velocity)	Continuous (2min)
Rainfall	0.2mm rainfall occurrence
REO, TSS, sediment deposition Contaminants*	Monthly (surface, sedi-ment bed trap, core samples)

Contaminants tested are: K, Pb, Fe, Cd, Cr, Mn, Mg, Al, Zn, Ca, Na, Ba, Cu, Ni, Sn .

Sampling Duration: 6 months

3.3 Detail of selected field sites

3.3.1 Campus field site

The swale is located within Heriot-Watt University grounds, Scotland. It is located parallel to a local road and collects runoff directly from this single camber road network via curb inlets. The swale has a mild gradient (less than 2%) over 100 m length, with well-maintained short grass (illustrated in Figure 3.4). This asset conveys stormwater runoff from approximately 500 m^2 contributing area where approx. 40% of land-use is impervious, urban developed area. The swale discharges to a piped stormwater network.

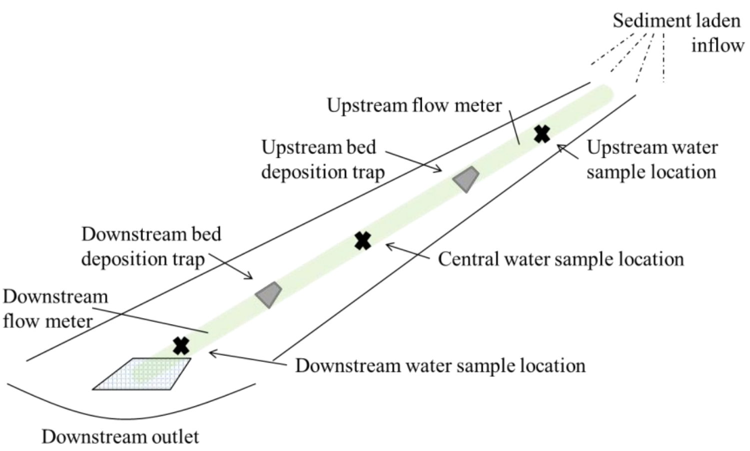


Figure 3.4 Schematic campus swale experiment diagram (Allen et al. 2015). Swale design comprised 40m linear downstream distance, discharging to a traditional outlet design (metal grate into vertical manhole chamber with lateral drainage pipe). During each of the three runoff events, water samples were collected every 5 minutes from the surface flow at three locations within the main flow path of the swale (1m, 20m and 39m from the upstream end, '**X**' annotation of the locations). Bed load traps were 300 x 300mm square inserted flush with the swale bed; these were weighted in place with gravels to provide the added benefit of reducing fine-particle wash-out post-capture. At the upstream and downstream (Figure 3.4), *Stingray* ultrasonic sensors were anchored on the swale bed and continuously logged flow depth and velocity. Post-experiment, a set of core samples of diameter 25mm and 0.02 m depth were taken at 5m intervals along the central flow path of the swale. Coring was then repeated at timeframes of plus 1 week, 6 months, and 12 months after the experiment.

The Campus field site was the focus of the REO tracer methodology trials only (Section 3.1). The sediment release of tagged material (REO tags: Nd and La), equated to ¼ of the annual average sediment loading, (annual loading defined from literature review

(Section 2.2) (i.e. 10kg). This was evenly spread over a 10-metre long by 1-metre wide strip of road upstream from the swale inlet. Wash-off was artificially generated using a pressurized local water hose for a 30 minute duration and equivalent to a three-month return period (3 month RP) runoff event. Use of a flow diffuser (replicated from Deletic 2004), located upstream of the tagged material, ensured sheet flow conditions across the road such that tagged sediment were routed towards the swale inlet. After a 1 hour drying period, this was repeated in a second and third artificial runoff event of equivalent duration and intensity; no additional sediment was released in these events.

The following monitoring was undertaken (with cross-reference to full methodology procedure):

1. *Surface water sampling* (Section 3.4.1), as shown in Figure 3.1 and 3.4. During each of the three runoff events, water samples were collected every 5 minutes from the surface flow at three locations within the main flow path of the swale (1m, 20m and 39m from the upstream end). Analysis of the surface water samples focused on identifying the particle size distributions in the suspended load, rates of transport and REO tagged sediment mobility.
2. *Bed deposit sampling* via sediment trap locations (Section 3.4.1), with data collected after each runoff event (see Figure 3.1 and 3.4). This data was collected to assess single-event detention efficiency and event-based transport within the swale, via bulk sediment analysis and assessment of REO concentrations.
3. *Sediment coring* (Section 3.4.1) was undertaken at the end of each experiment to quantify the cumulative sediment detention over multiple flow events and time. Coring was then repeated at timeframes of plus 1 week, 6 months, and 12 months after the experiment (Section 3.5.6). Focus was placed on analysis for longer term REO detention efficiency of the swale, in measureable quantities.
4. *Flow monitoring* (Section 3.4.4) was undertaken continuously within the swale for the experimental period (during the three artificial flow events) using a *Stingray* depth/velocity/flow meters. Analysis of the flow data provides velocity, depth and discharge event information specific to the drivers of sediment movement, deposition and detention within the swale.

ICPMS analysis (methodology presented in Section 3.6.5) was completed for REO concentration analysis of all samples, runoff event surface samples, bed deposition and core samples, were analysed using an ICPMS. Background "control" samples were also analysed for REO concentration of both the artificial water supply and local soil samples. Full details of the ICPMS methodology are provided in Section 3.6.5, with the intention of these data to assess the potential use of REOs as tracers in SuDS networks. The results of these experiments are provided in Section 3.6.6.

3.3.2 J4M8 field site

The J4M8 distribution park (located in Bathgate, Scotland) incorporates a set of established and well maintained SuDS treatment train networks. This commercial area has been designed as a 'pipe-less' development, conveying all stormwater (for the ~ 0.3km^2 site, of which less than half falls to the monitored SuDS networks) via vegetated surface measures to the legal point of discharge, the River Almond. The SuDS assets within J4M8 comprise of vegetated filter strips (VFS), vegetated swales, linear wetlands, a wetland and a pond (Figure 3.5) and seek to treat three identified land-use typologies (car park, road and roof).

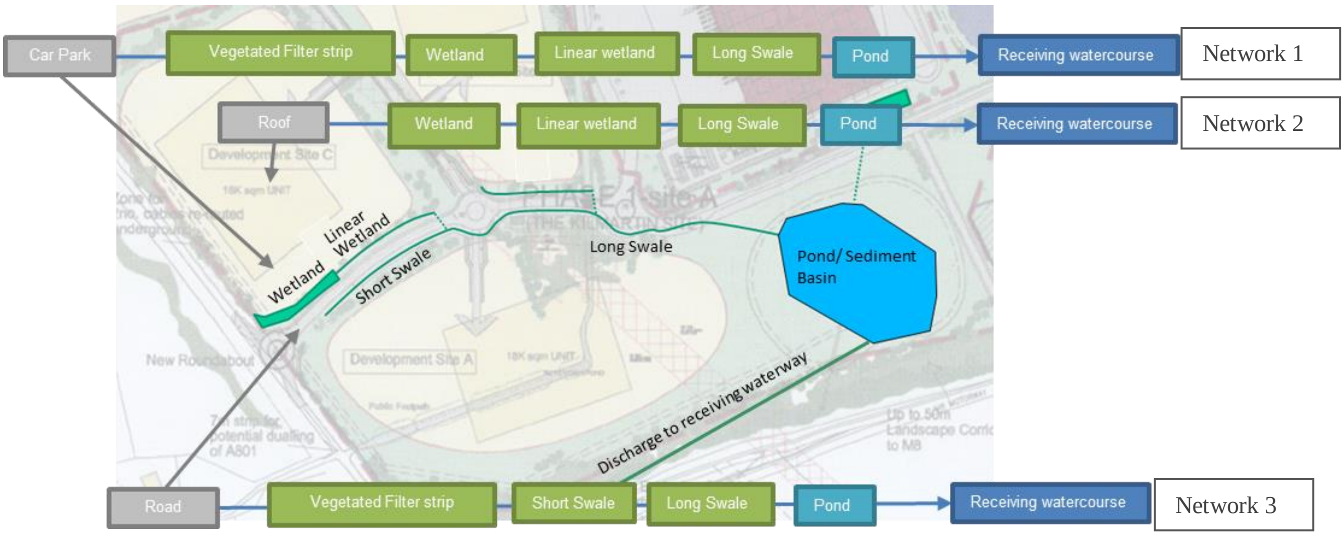


Figure 3.5 Schematic of J4M8 SuDS network and key urban pollution surfaces which route surface waters into the River Almond.

Fine sediment was tagged using unique rare earths (REO), detailed methodology presented in Section 3.6. Tagged sediment was released from three specific locations:

on a specific area of car park within the distribution centre, within the downpipe from the roof runoff of the distribution centre building, and on the internal road surface (indicated in Figure 3.5). Sediment, equivalent to 1/12 of the annual sediment pollutant load for this urban area, was tagged using rare earth element tracers (tagged and released on site every 3 months). Three separate sediment volumes were created, for release onto the separate car park, roof and road locations, each using a unique individual rare earth tracer. The REO tracers used for the car park, roof and road were Nd, Sm Gd, Tb; Y, La, Ce, Pr; Dy, Ho, Er, Yb respectively. Tagged sediment, once released, was left to move naturally off the urban surface (roof, car park or road) via rainfall-runoff events, into and through the SuDS network.

The following monitoring was undertaken (with cross-reference to full methodology procedure):

1. *Surface water sampling* (Section 3.4.1) was collected across the wetland, linear wetland and swales shown in Figure 3.5 (asset specific sample locations are depicted in Sections 5.3-5.6). Samples were collected from within the main flow path. Surface water samples provided information on the quantity of suspended sediment and solute urban pollution within the fluid component of the SuDS asset(s) at the point in time of sampling, this informing the mass balance for the asset(s).

2. *Bed deposition sampling* (Section 3.4.1) was conducted using sediment traps placed below the surface sample locations. Sediment traps were designed, using Van Rijn (1984) saltation assessment, to ensure material up to 2mm in particle size were collected over the two week sampling period. Sediment traps were set into the bed of all SuDS assets, maintaining a level bed surface where sediment traps were located. Sediment trap data was analysed to identify the (fortnightly) deposition within each asset, to consider medium term (fortnightly) deposition rates relative to rainfall and flow characteristics.

3. *Sediment coring* (Section 3.4.1) was undertaken within 1m of the sediment trap, with no two samples occurring at the same location. Cores were collected to provide time stamped mass deposition of (tagged and un-tagged) sediment within each of the SuDS assets. This data provided temporally specific mass and

tagged sediment deposition information, allowing analysis of asset detention via deposition efficiency within the SuDS asses from bulk sediment analysis and assessment of REO concentrations in the core sample material.

4. *Rainfall and flow monitoring* (Section 3.4.3, 3.4.4) occurred continuously throughout the monitoring period. Rainfall data was collected on site, flow was monitored within the wetland, linear wetland and swale. Collated, this data permitted detailed analysis of sediment deposition potential, distribution, residence and flushing efficiency for both individual SuDS assets and the whole system

Surface water sampling and bed deposition (sediment trap and core sampling) were monitored fortnightly over 12 months to provide a fine resolution (temporal and spatial) dataset of multiple runoff event sediment transport. The sampling interval was specifically designed to capture as many sample points as physically and economically viable over a 12 month period. Daily sampling would have provided a more detailed dataset but at the cost of a higher fine sediment and REO trace removal. Monthly sampling was considered too course a time step, with a higher likelihood of the REO tagged sediment passing without detention in the traps of surface flow samples. Therefore, given the economic and physical time constraints on sampling, the fortnightly sampling regime was adopted with acknowledgement that a smaller sampling time step may provide more detailed results.

ICPMS analysis (Section 3.6.5) was completed for REO concentration analysis of all surface, sediment trap and core samples. As with the Campus swale, background “control” samples were also analysed for REO concentration of the SuDS asset surface water, bed deposition and surrounding land use surfaces (road, car park, roof) and topsoil material to ensure background concentrations were negligible, thus ensuring REO use as a sediment tag was appropriate. ICP-OES analysis (Section 3.5.3) was completed on sediment core samples (background and experimental samples) and urban surface samples (roof, road and car park). ICP-OES analysis provided background and time stamped (quarterly) urban pollutant concentration results (presented in Chapter 7) to identify the level of pollutant concentration in individual SuDS assets and on the case study urban surfaces.

3.3.3 NGP field site

Newcastle Great Park is a new high intensity urban development area that is planned as part of the greater urban expansion of Newcastle City. The development site includes sustainable drainage measures, specifically SuDS ponds, for water quality treatment and stormwater runoff quantity mitigation.

The development area pertinent to this field site is approximately 1.32km^2. This development area is directly connected to a downstream SuDS pond located on the Ouseburn floodplain. The contributing catchment area is comprised of a community centre, primary school and predominantly semi-detached and flat apartments. At the time of the field experiments, approximately 60% of the site was developed, with further urbanisation occurring across 30% close to the pond.

The urbanised area is intensely developed, with an impervious area of 80% comprised of impermeable road, car park, paving and roof area. All stormwater from the developed area is collected and conveyed via road inlets and pipes to the stormwater pond. There are no upstream or source control SuDS measures within the contributing catchment or stormwater network.

Figure 3.6 presents a schematic illustration of the sample locations and flow monitoring points within the NGP SuDS pond. This pond not only treats stormwater from a high intensity urban development area, but includes the influence of a proportion of construction runoff. This makes this pond a useful addition to the J4M8 and campus swale field sites, allowing for extension of the urban land use impact assessment. The pond within J4M8 is large, approximately 16,240m^2, with a depth of greater than 0.5m.

The NGP SuDS pond is significantly smaller in area, 2,400m^2. The average pond depth is below 0.5m and thus it is physically possible to wade across this pond to undertaking survey and sampling activities. Sampling locations were located within the pond open water section, reeded area and backwater section, illustrated in Figure 3.6.

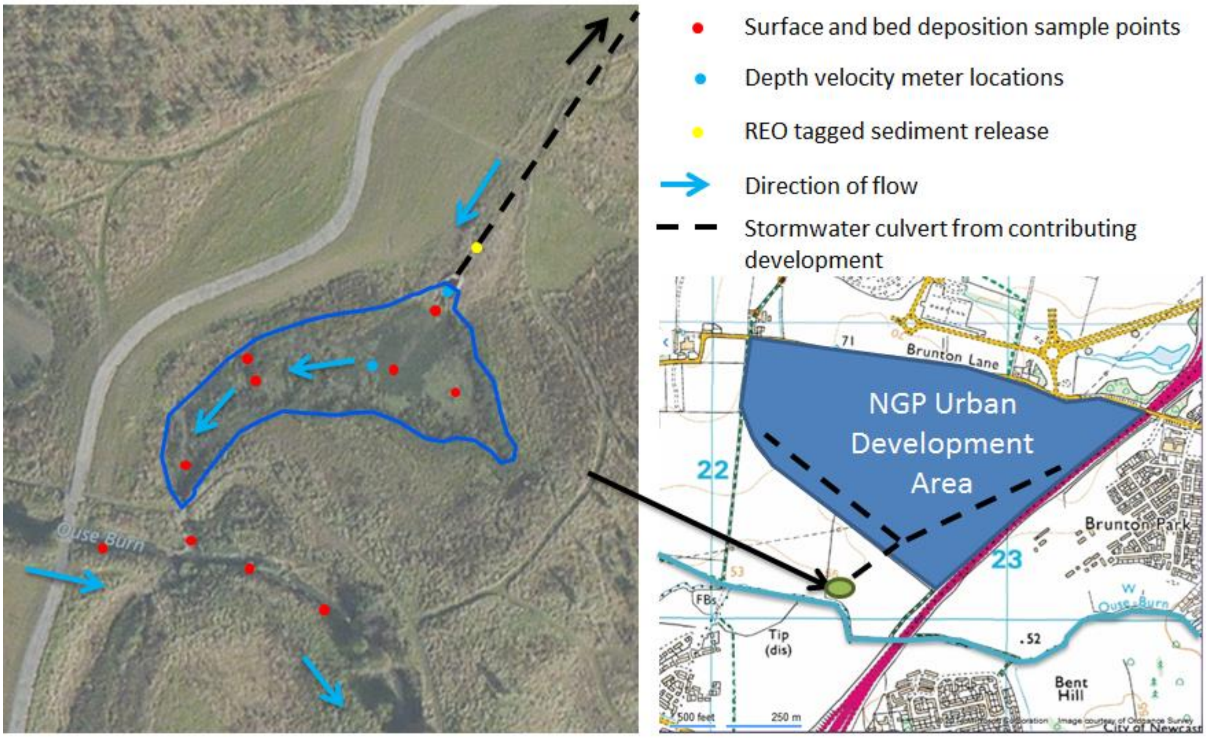


Figure 3.6 Schematic of NGP pond experimental set up and sample locations. Surface flow, sediment trap and core sampling locations are indicated in red, with flow monitoring locations indicated in blue. Tagged sediment was released into the inlet pipe close to the discharge into the pond. The stormwater and receiving waterway flow paths are indicated by blue arrows.

The following monitoring was undertaken (with cross-reference to full methodology procedure):

1. *Surface water sampling* (Section 3.4.1) was collected at the pond inlet, 5 locations across the pond, upstream and downstream of the pond outlet and within the receiving watercourse upstream and downstream of the pond outlet, Figure 3.3 and 3.6. Surface water samples analysis identified the suspended sediment (both tagged and un-tagged) concentrations across the pond, thus informing the sediment mass balance analysis.

2. *Bed deposition sampling* (Section 3.4.1) was conducted using sediment traps placed adjacent to the surface sampling locations. Sediment trap design and implementation followed the same method as used across J4M8 field site. Sediment trap mass deposition was used to inform the fortnightly deposition rate at specific locations across the pond, while REO tagged sediment concentrations

of deposited material provided a fortnightly accumulation indication relevant to the fortnightly rainfall/flow analysis undertaken in Chapter 6.

3. *Sediment coring* (Section 3.4.1), as with the J4M8 sampling regime, was undertaken within 1m of the sediment trap. This data was analysed to illustrate the change in REO tagged sediment deposition at each of the monitoring locations over the monitoring period, in conjunction with mass deposition (non-tagged sediment) input into the mass balance analysis. Deposited sediment urban pollutant concentrations were also analysed to identify the level, and potential hot spot, contamination within the pond.

REO tagged sediment was released at the pond piped outlet. Sand sized material (d_{50} 225 µm) was tagged with Gd, Yb, Er and Dy; silt-clay sized material (d_{50} 28 µm) was tagged with Go, Pr, Sm and Nd REO tags. The first release occurred at the end of January 2015, with 3 further sediment releases at 4 week intervals after this. Stormwater inflow into and within the SuDS pond was continuously monitored using a *Stingray* depth velocity metres (Section 3.4.4).

Sediment samples of the surface flow and bed deposition were undertaken monthly (prior to each new sediment release). Due to the location of the pond from the University, only monthly field visits were financially viable. Surface samples were collected from adjacent to the sediment traps inset into the SuDS pond bed. This allows surface samples to be collected from the same location and in the same manner at each sample period. The NGP pond was monitored for a period of 6 months.

3.4 Field sampling methodology

For all sites in Section 3.2, sediment was monitored within: (a) the surface flow, to assess suspended sediment load, particle size distribution and REO/pollutant concentrations; and (b) bed deposits to analyse the detention efficiency, particle grain size distribution and REO/pollutant concentration. For all case studies, fine sediment samples were taken from known (spatially marked) locations throughout the SuDS assets at regular time intervals. Specific details of methodological procedures are provided below.

3.4.1 Surface flow and bed sediment samples

Ephemeral Assets: Flow samples from swales and linear wetlands were only viable when surface flow reached a depth ≥20mm. In all cases, sampling employed a 500ml open-neck container submerged within the upper 20mm of flow depth for a period of 60 seconds; this surface sampling is fit-for-purpose in terms of gaining a general understanding of runoff loadings where sediment particle size distributions are small.

Perennial Assets: For continuously wet SuDS assets, samples were collected using an *ISCO 3700* sampler sited in the main flow path of the asset. This permits water sample collection at a consistent pumped rate (approx. 60 seconds after flushing) through flushed sample hoses in both the surface water and near-bed flow regions (Figure 3.7). To ensure repeat sampling at identical locations, hoses were anchored to a selected, representative location in the pond and wetland. Surface sample (Figure 3.4) collection employed floatation devices to ensuring sampling depth within the upper 50mm of the water column, independent of time-dependent variability in the actual water depth within the asset.

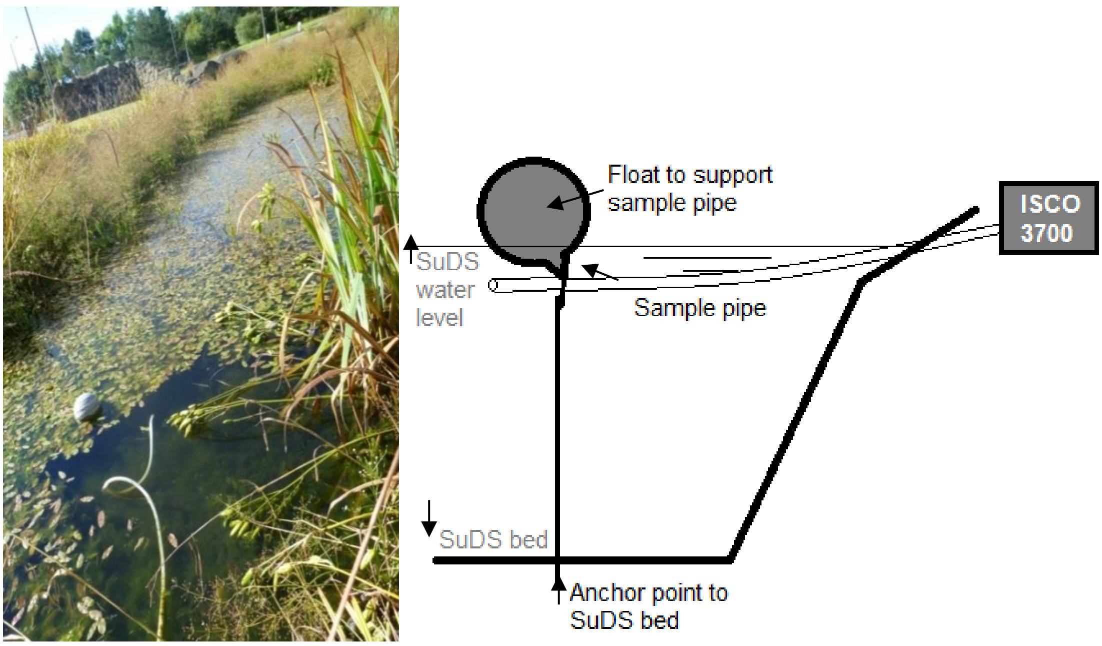


Figure 3.7. Sketch of sample hoses and floats for perennially wet SuDS asset sampling. Hoses were 10mm diameter. Hoses were flushed before and after each sample occurrence. The hoses were placed in the wet assets as static sampling measures to ensure samples were collected from the same location and depth at every sample occurrence.

Bed deposition sampling

Bedload traps: Bed deposit samples were collected in sediment traps (Figure 3.8) immediately below the surface sample locations. Each trap comprised of two containers, an outer box permanently anchored into the bed and remaining in place for the total period of sampling and an inner box (the sediment trap) which was removable for sampling of the collected deposits. Each trap comprised a 300x200mm surface area opening fitted flush to the bed surface (to avoid disturbance of active transport processes) with 5L volume and depth 100mm (appropriate to permanent capture of transported bedload), similar to that used in Kayhanian et al (2012). Traps were weighted in place with 500g of 100mm diameter gravel, used to provide the added benefit of reducing fine-particle wash-out post-capture; the fine particle load of interest was separated from the gravel media by basic sieving (Figure 3.8). This design is common in watercourse studies (e.g. Lawler 2006) as retaining the outer box during trap removal/emptying minimises disturbance of the substrate. In the present thesis, a very small amount of highly localised artificial resuspension, persisting for < 2 minutes, arose from suction/turbulence processes in trap removal; no disturbance was noted during trap replacement. Observations 15 minutes after trap replacement recorded no visible sediment accumulation on the gravel surfaces; this provides a degree of reassurance that potential overestimate of fine sediment deposition due to sampling error is minimal (compared to the measured sample load). Overall, the temporal frequency of sampling was set specific to giving an indication of the short-intermediate term sediment deposition; whilst the Campus swale (Section 3.3.1) was monitored post-event to test REO methodology protocols, the longer-term studies of J4M8 SuDS and NGP pond (Sections 3.3.2-3.3.3) considered a 2 and 4 week monitoring cycle.

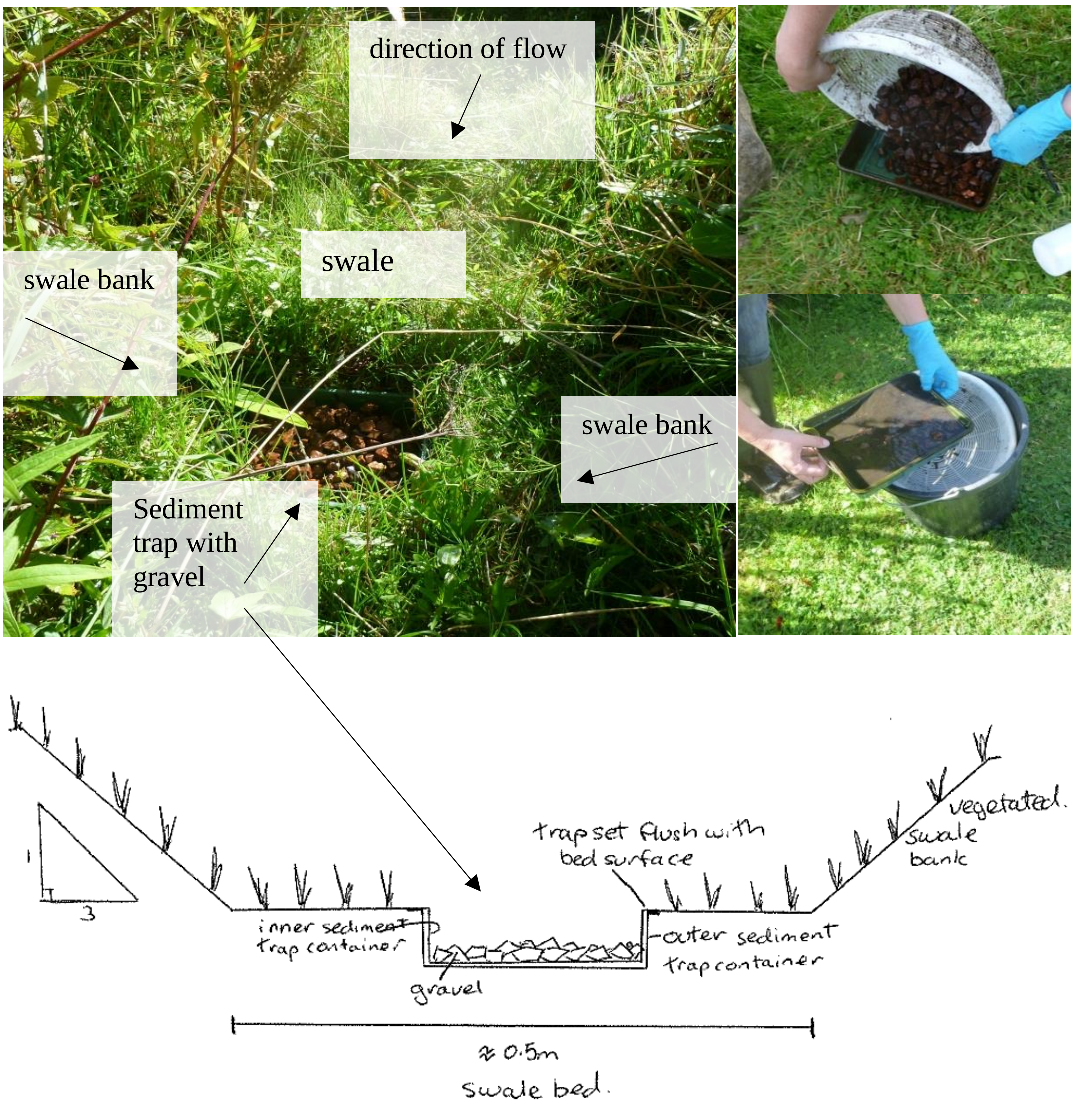


Figure 3.8. Sketch of sediment traps in the field. Left hand photo shows the sediment trap flush with the swale bed. Right hand photos show the sediment trap full (bottom) and being reset with clean gravel (top). Similar designs of trap (300x200x100mm) are also used in e.g. Lawler 2006, IAEA 2003, Fraley 2004.

Core samples: Core samples employed a design similar to that used in IAEA (2003) and Fraley (2004), and were taken from small surface areas adjacent (within 1m) to each of the sediment traps (J4M8 and NGP); the specific location varied for each sample to ensure cumulative deposition was represented accurately. Samples of 50mm diameter were collected to a maximum depth of 100mm; depths were variable and dependent upon the lining depth of the asset design (e.g. the campus swale liner was only 20mm sub-surface, Section 3.3.1). Coring frequency for J4M8 and NGP sites corresponded directly with the bed load sampling regime and was designed for identification of the cumulative multiple event REO tagged sediment deposition and detention. Subtle distinction in the methodology applied in the Campus swale tests (Section 3.3.1) was coring intervals at 5m along the asset taken immediately post-experiment, +1 week, +6 months, and +12 months; this was specific to spatially-intensive investigation of the REO methodology for long-term measurements.

3.4.2 Urban source sampling

Across all field sites, existing urban sediment (natural and anthropogenic) was sampled to identify source- and site-specific fine sediment loadings and particle size characteristics.

Availability sampling: Samples were collected after a dry period, defined as 48 hours without rainfall-runoff occurring. Employing a 1m x 1m frame weighted to the surface, the internal area was subjected to manual brushing to loosen surface deposits as appropriate for vacuum-sampling (Vaze et al. 2002, Egodawatta and Goonetilleke 2006). This representative $1m^2$ of surface material was dried, weighed, sieved and analysed for particle size using traditional phi-scale sieves (coarser fractions >2mm) and the *Mastersizer S* (finer fractions <2µm); see also Section 3.5.1. To ensure representativeness, triplicate samples were collected from each dominant source type within the case studies; i.e. Campus swale sampled the road only; J4M8 sampled car park and road surfaces; NGP sampled residential, construction roads and car park surfaces (Section 4.2.1). Roof surfaces were not considered due to 'working-at-height' health and safety constraints.

3.4.3 Rainfall monitoring

Rainfall was monitored at the Campus field site using an existing *Davis Vantage Pro 2* with 0.2mm tipping bucket design, with a logger that records rainfall continuously. This located on the main campus building, away from vegetation effects yet within 500m of the test swale. An equivalent system (*Tempcon RG3-M*) was deployed at the J4M8 field site located immediately adjacent to the wetland, elevated 2m above the land surface (Figure 3.2). Both the Davis [1] and Tempcon [2] rain gauges are calibrated by the manufacturer prior to field installation.

[1] Davis Instruments Installation Manual, www.davisnet.com (last accessed 08/10/2016)

The NGP rainfall information was collected from the local Meteorology Office rain gauge (0.2mm tipping bucket gauge) located at the Albemarle Barracks, Newcastle (10km west from NGP) (data made available by FOI). The details of the equipment used to collect the Newcastle rainfall information (gauge make/design) were not provided by the Meteorology Office, but the rain gauge information was collected by a calibrated rain gauge and was validated in accordance with Met Office data standards. The Meteorology Office rain gauge was used for the NGP field site rainfall information due to the close proximity within a level geographical location.

Within this research, a rainfall event has been defined as the occurrence of continuous rainfall to a depth ≥ 0.4mm depth. A cluster of rainfall events is defined as 2 or more rainfall events occurring within 24hours of each other.

3.4.4 Stormwater depth/velocity monitoring

The depth and velocity of stormwater flow within the monitored SuDS assets was collected using a field-portable Greyline *Stingray*[3] depth/flow/velocity meter. Via doppler sensing (*QZ02*) data on depth and velocity is logged at an interval of 10 seconds to 20 minutes. Water depth/velocity was logged every 2 minutes at all sites (enabling 4 weeks of logging capacity while ensuring the significant changes in flow conditions were logged). Water level depth measurements offer accuracy 0.25% of the range 25.4mm to 4.5m, whilst velocity is accurate within ±2% between 0.03-3.05m/s. This is appropriate to the field sites where monitored flow depths ranged from 0.02m - 1.16m and velocity ranged from 0.1m/s – 0.79m/s (Section 4.3.1). The sensors were placed on the base of the SuDS asset, anchored in place using manufacturer supplied case and left to continuously monitor over the full monitoring period(s). All equipment was calibrated by the Manufacturer prior to deployment, and tested within the laboratory (static depth and known velocity tests) to confirm calibration.

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2 Tempcon Instrumentation Inc. User’s Guide Installation and Operation Instructions, www.tempcon.co.uk (last accessed 08/10/2016)

3 Users Guide’ Stingray 2.0 provide by Greayline Instruments Inc., www.greyline.com , (last accessed 08/10/2016)

3.5 Non-REO trace laboratory analysis methodologies

The following section presents the methodology for the laboratory based sample analysis undertaken on the field collected sediment samples of Section 3.3.

3.5.1 Particle Size Distribution (PSD) analysis methodology

The purpose of the particle size distribution analysis was to identify the modal and d_{50} characteristics of suspended and deposited sediment within the field samples. While it is acknowledged that urban sediment (fine sand to clay material) has cohesive properties and thus can form cohesive particulates, the focus of this research does not extend to delineation between cohesive and non-cohesive particle size analysis nor the PSD analysis of compound (cohesive) particulates. As such, the particle size analysis has been undertaken following standard practice and therefore provides dispersed PSD results for field samples.

PSD testing was undertaken using two methods; sieving and laser diffraction. Field samples were dried at 105 °C for 24 hours. Dried sample material was then manually sieved; in line with the objectives of this thesis fractions $\leq$ 2mm were retained for analysis (4.2 µm up to 2000 µm) using the laser diffraction method of the *Mastersizer S* (long bench configuration). For this latter method, fines were re-suspended in deionised water via shaking for up to 5 minutes. The suspended sediment sample was added to the sample pump and passed through a laser beam; detectors placed at fixed angles measure the intensity of light scattered at that position. A mathematical model (i.e. Mie/Fraunhoffer Theory) is then applied to generate a particle size distribution (Etzler and Deanne 1997, Ramaswaamy and Rao 2006); the final result is reported on an Equivalent Spherical Diameter Volume basis. It is important that acceptable sample obscuration values for this method are below 30%; thus, any samples with a greater obscuration were diluted further to ensure accurate Mastersizer results. All samples were tested in triplicate, all samples providing results within the acceptable variance of ±5%. The sample d_{50} and unimodal particle size were recorded for each sample.

3.5.2 TSS and mass bed deposition analysis methodology

Total suspended solid analysis was undertaken following the British Standard guidelines, BS EN 872:2005. The total volume of each collected sample was recorded for both surface samples. *Whatman* filters (0.45 µm pore size) were prepared for sampling by saturation with de-ionised water, followed by drying at 105 °C for 24 hours. The saturated and dried filters were then weighed using a gravimetic scale (accuracy to 4 decimal places- grams) and the initial filtration weights recorded. Then, each field sample was thoroughly mixed via 2 minutes of consistent aggitation. 20ml of suspended sediment sample solution was subsequently filtered through the filter paper with the aid of a vacuum pump. After filtration, each sample was returned to the oven for drying, at 105 °C for a further 24 hours. The dried residual sediment and filter was then weighted using the same gravimetric scales and the sediment + filter weights recorded. This increase in filter weight was used to calculate to Total Suspended Solids (TSS) in the surface samples, providing a concentration of suspended sediment per sample volume (scalled to conventional mg/l (ANZECC 2000, Li et al. 2007, Fraley 2004)) to allow the calculation of suspemded sediment in SuDS asset stormwater (volume) at the sampling point in time (applicable to the mass balance analysis). All samples were completed in triplicate and presented results within the acceptable ±5% range.

To calculate the deposition mass, the total mass of material detained within the sediment traps and the core samples were also calculated by weight. Deposited sediment sample material was dried for 4 days (at 105 °C) and then weighed. This provided the total mass (kg) of sediment material collected in the sediment traps (300x200mm surface area) and from the core samples (25 and 50mm diameter).

3.5.3 Heavy metal analysis methodology

Bed deposition sediment samples collected within the J4M8 and NGP SuDS were analysed to provide information on the pollutant levels Section 2.2. evidences previous studies (e.g. Zander, 2004; Li et al., 2006; Kayhanian et al., 2012) defining the urban particle size distributions relevant to the transport of dominant metal pollutants; based on this guidance, Section 3.4.1 notes that field samples were sieved truncated at an upper threshold of 2mm grain diameter for the purpose of this thesis' analysis. Pollutant concentration analysis of the remaining fine sediment was undertaken using an

Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES). Sample preparation followed the seven-stage standard HNO_3-H_2O_2 acid digestion method (EPA method 3050B) (Nham 2010) (also outlined in BS ISO 13547-2:2014 and used by Hseu (2004)). The acid digestion preparation steps are listed in Appendix II.

The standard array of 15 trace elements and urban pollutants assessable by ICP OES were tested; K, Pb, Fe, Cd, Cr, Mn, Mg, Al, Zn, Ca, Na, Ba, Cu, Ni, Sn. ICP OES was also used to quantify the total P nutrient concentrations within each sample. Due to the limitations of this study (experimental and financial) organic matter analysis was not included in the pollutant analysis. However, it is noted that organic matter potentially has a significant influence on cohesive flocculation and particle aggregation and pollutant adsorption (Schorer 1997). Results were provided in part per million (as appropriate to the quantity identified). As part of this procedure, the ICP-OES sample analysis calibration confidence and tolerance intervals are provided in Table 3.2 and standard calibration (against benchmark calibration solutions) prior to sampling was performed by the Scottish Universities Environmental Research Centre (SUREC).

Table 3.2 ICP-OES analytes confidence and tolerance intervals. Data provided via academic related technical services standards for the SUERC ICP-OES facility used in this thesis. All values are in parts per million (ppm). The ICP-OES was calibrated for pollutant concentrations of 0.1ppm to 10ppm.

Analyte	Result range in this thesis (min – max)	Consensus value (calibration)	Consensus value Confidence interval	Consensus value Tolerance interval
K	0.1 - 3.1	3.19	3.13-3.25	2.76-3.63
Pb	0.1 – 10.8	0.112	0.111-0.113	0.101-0.0123
Fe	0.1 – 1.7	1.29	1.27-1.30	1.17-1.41
Cd	0.2 – 2.16	0.224	0.221-0.226	0.203-0.244
Cr	0.1 – 9.80	0.437	0.432-0.443	0.394-0.481
Mn	0.1 – 2.0	0.345	0.342-0.349	0.318-0.373
Mg	0.1 – 2.0	6.92	6.83-7.00	6.31-7.52
Al	0.1 – 2.4	0.233	0.29-0.238	0.199-0.268
Zn	0.1 – 2.3	0.881	0.781-0.891	0.809-0.954
Ca	0.1 - 1.9	7.69	7.59-7.79	7.00-8.38
Na	0.1 – 2.4	19.1	18.9-19.3	17.4-20.8
Ba	0.2 – 1.6	3.35	1.72-1.77	1.55-1.94
Cu	0.3 – 2.6	0.844	0.833-0.856	0.757-0.931
Ni	0.1 – 6.1	0.840	0.830-0.849	0.767-0.912
Sn	0.3 – 5.7	0.916	0.897-0.934	0.789-1.043
P	0.1 – 3.0	0.495	0.485-0.505	0.432-0.557

3.6 REO tracer methodology

3.6.1 Rationale and novelty

The aim of the sediment transport monitoring field work was to create a novel dataset on sediment supply to and transport through the SuDS system as a result of numerous consecutive events. Section 2.8 summarises both the range of sediment tracing methods available and provides detailed discussion of the benefits and constraints of the more frequently employed tracer techniques. As tracer use for sediment movement in SuDS is novel, the present thesis' dataset is unique and methodology specific for SuDS application was devised as a core objective of the present research. With regard to best-practice, the non-SuDS evidence base has been reviewed against a number of important requirements (Criteria) specific to tracer requirements; specific to the SuDS system, these include:

1. *Persistence* - sediment supplied to the upstream boundary should be tagged via a conservative tracer persisting (adsorbed to the sediment) over a range of spatial (m-km scale) and temporal (up to 12 month) scales.
2. *Non-toxic* - as the blue-green drainage network has environmental value and importance, the tracer must result in no detrimental impact on the receiving environment.
3. *Particle characteristics* - the tracer must be effective in mimicking natural sediment movement; hence, particle size, grain size distribution and density must be considered
4. *Multiple signatures* - several unique forms of tracer must be available to enable the monitoring of individual sediment releases (time-lapse repetitions and source-specific identifiers) within the same receiving SuDS.

The currently available and documented sediment trace methods have been outlined in Chapter 2 (Table 2.9). Review of these methods identified that six (6) techniques that may be of merit. However, after imposing the tracer criteria outlined above four of these methods fail to the requirements. *Pollen* and *magnetic fluorescent* material tracers are limited in availability precluding compliance with Criteria 4; pollen by the natural availability and fluorescent particles by the artificial fluorescent colours available. *Painted natural particles* have limited field resilience, breaching Criteria 1.

Radionuclide (Sabourin and Wilson 2008, Deletic, 2001) tracers have been recorded to move both adsorbed and without adsorption to sediment across natural surfaces (Parsons and Foster 2011) and, in many locations require environment agency permission for use, thus limiting method adoption and failing Criteria 1 and 2.

Once these four techniques had been removed from the decision making process, a more detailed evaluation of the remaining two methods (finger printing and REO) was undertaken.

Fingerprinting has been demonstrated to be an effective watershed erosion and sediment (Zhang et al. 2001, Zhu et al. 2010) transport tracing method using naturally occurring periodic element concentrations and particle size distribution to determine a sediment source. Where the range of sediment sources have distinctly different signatures, for example forestry erosion *versus* urban wash off, the fingerprinting method is effective. However, sediment entering SuDS are derived from road, car park and roof surfaces all within an urban area developed areas. While the particle size and heavy metal concentrations do differ between these sources, the source specific urban signatures are not as easily discernible as those from the far larger scale catchment studies that this technique has been previously used for. Further, this method cannot provide time-stamped supply information at the resolution required for urban studies, thus failing Criteria 4 of the selection process.

Rare Earth Oxides (REO) were, therefore, the only tracer method which complied with all selection criteria. There are 17 trace signatures available (UK background levels are low, at the limit of analysis) 15 of which are easily analysed. These 15 signatures permit individual source and time-stamped identifiers. REOs adsorb easily to natural sediment, ensuring that grain size characteristics of source material can be preserved, and have shown very limited field detachment in laboratory testing (Zhang et al. 2001, Zhu et al. 2010). In addition, the method has been successfully used in agricultural scour and erosion research providing a basis for methodological development appropriate to SuDS application. For this purpose, it should be stressed that the methodological refinement, testing and application of REO tracing in SuDS holds three degrees of novelty: (i) to date, REO tracing has not been undertaken through an urban drainage network; (ii) fine sediment trancing of multiple event movement has not been undertaken in ephemeral or urban perennial flow paths (iii) multiple unique signatures have not been used to trace fine urban pollution from source (urban surface) to sink (deposition zones in SuDS). In summary, this method permits the following:

1. multiple unique identifiers that can attach and trace sediment from 8mm to 0.45 µm in size;
2. tagged sediment to move naturally;
3. monitoring for months/years as the trace is effective and stable in the urban drainage environment for an extended period without degradation;
4. fine sediment transport monitoring in a vegetated environment as the trace is not taken up by plants, does not react significantly under aerobic or minor acid fluctuations and does not become released and reabsorb to passing sediment material;
5. negligible environmental or aesthetic impact (very small amount of trace is necessary to monitor a large sediment mass movement); and,
6. experiments to be repeated within the same test environment (SuDS) without interacting or modifying ongoing/future experiment results.

As appropriate to methodological development, the REO tagging methodology underwent an iterative process of literature → methodological refinement → pilot test → evaluation in tightly-controlled environmental simulations of Campus swale site (Section 3.3). Upon qualification that the method was fit for purpose, the REO tagged sediment methodology was then implemented across the J4M8 and NGP field sites (Figure 3.9).

Figure 3.9 is a schematic of the REO methodology and respective field experiments that have been undertaken as part of this thesis research and are discussed within this methodology section.

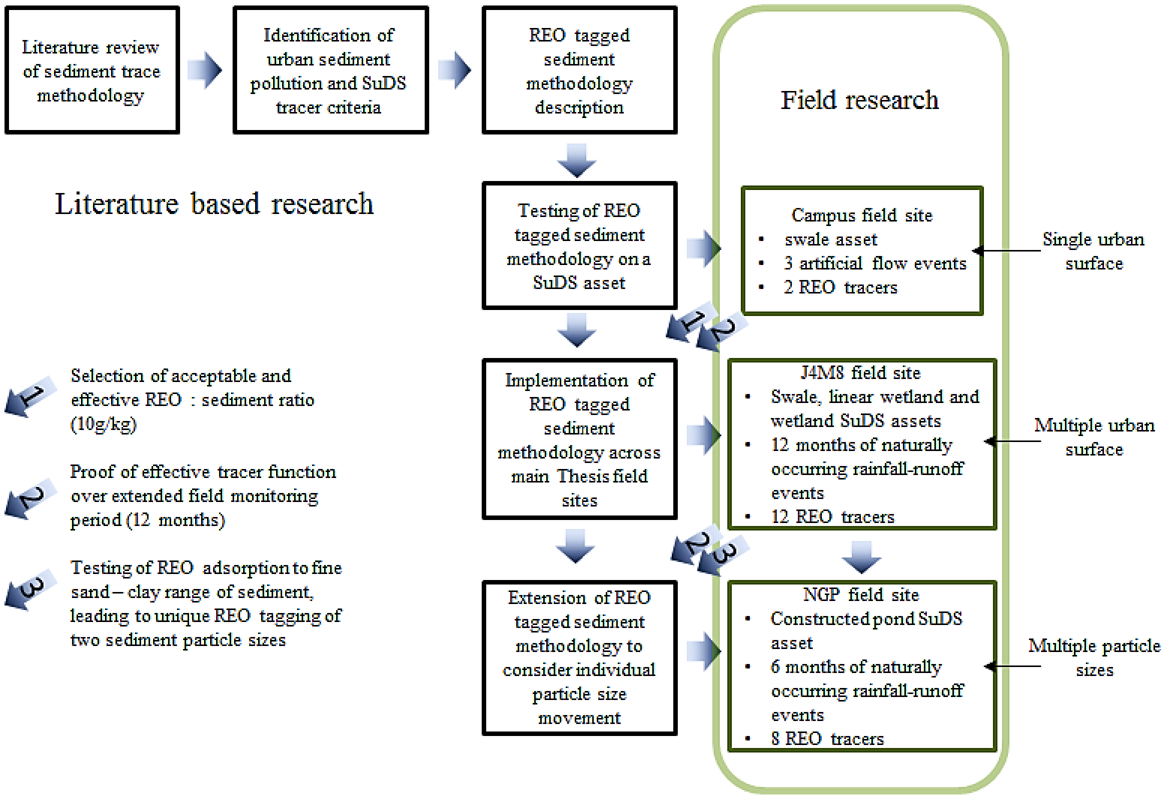


Figure 3.9 Schematic of REO tagged sediment methodology evolution and implementation. Numbered arrows illustrate the linkage between field experiment findings and a change in the REO tracer methodology.

3.6.2 The REO tracer use on fine sediment

Rare earth oxides (REO) are naturally occurring elements within the earths surface, naturally occurring in very low concentrations (<2ppm in the bulk Earth) resulting in the description *rare* (rather than limited spatial abundance) (Adachi et al. 2004). REO are strongly electro-positive (+3 valencies) and become adsorbed to clay, silt and sand material through an ion-adsorption process. REO adhere to sediment creating a (partial) film on the surface of the sediment material by creating an ionic bond through physisorption (Henderson 1984, Papangelakis and Moldoveanu 2016). Physisorption employs van der Waals force/bonding, creating chemically weak bonds between the REO and urban sediment. Table 3.3 presents a summary of the REO (used in this research) characteristics.

Table 3.3 REO characteristics, for REO used in this thesis (Adachi et al. 2004, Zepf 2013, Henderson 1984)

Rare Earth Element	Atomic number	Orbitals/ Electronic configur-ation	Orbital states	Atomic Radius (pm)	Ionic Radius (pm)	Atomic Mass (u)	Boiling point (k)	Melting point (K)	Density (kg/m3 at 293K)	Isotopic half-life (yrs)
Ho	67	$4f^{11}6s^2$	+3	176	90	164.9	2968	1747	8795	'stable'
Pr	59	$4f^36s^2$	+3, +4	182	99	140.9	3785	1204	6773	'stable'
Sm	62	$4f^66s^2$	+3	181	97	114.9	3000	1441	7220	$1.06x10^{11}$
Nd	60	$4f^46s^2$	+2, +3, +4	181	98	144.9	3341	1294	7007	$2.1x10^{15}$
Gd	64	$4f^75d^16s^2$	+3	179	94	157.3	3539	1586	7900	$1.1x10^{14}$
Yb	70	$4f^{14}6s^2$	+2, +3	193	87	173	1466	1097	6965	'stable'
Er	68	$4f^{12}6s^2$	+3	175	89	167.3	3136	1802	9066	'stable'
Dy	66	$4f^{10}6s^2$	+3, +4	177	91	162.5	2835	1685	8550	$2x10^{14}$
Tb	65	$4f^96s^2$	+3, +4	178	92	158.9	3396	1629	8229	'stable'
Y	39	$4d^15s^2$	+3	180		88.9	3911	1795	4469	'stable'
La	57	$5d^16s^2$	+3	187	103	138.9	3730	1194	6145	$1.05x10^{11}$
Ce	58	$4f^26s^2$	+3, +4	182	101	140.1	3699	1204	6773	'stable'

Note: ‘stable’ is geochemically defined as theoretically capable of spontaneous fission, thus not considered to decay.

REO have slight differences in electronic configurations (Table 3.3, Orbitals/Electronic configuration column) but the different configurations are not significant enough to result in different chemical behaviours (Henderson 1984). Thus, the ionic bond formed during adsorption is similar in strength for all REO. REO isotopes (commercially available and used in this study) are either stable or have long half-lives, illustrating that tracer decay and maturation is unlikely to cause unexpected shift in tracer efficiency (Henderson 1984, Zhang et al. 2001, Polyakov and Nearing 2004). Environmental change, such as acidification, is considered more significant elements of potential REO tracer limitation, identified in Kimoto et al. (2006) to be 4% and Zhu et al. (2010) review of up to 15%. Review of REO tracer use in soil erosion published research identifies few leaching or maturation studies, with Zhang et al. (2001) and Kimoto et al. (2006) being well cites as indicators of REO tracer efficiency. Within both these studies REO leaching experiments were undertaken using column testing over 20-40 days. Results show negligible change in REO tracer validity or efficiency; a 4% enrichment potential is reported in Kimoto et al. (2006). However, a significantly longer study has been undertaken by Polyakov et al. (2006) of 16 months (over 80 moderate rainfall events). During this study no significant maturation or decay in REO trace efficiency was noted. This provides a published sediment tracer experiment of greater duration that the thesis study and some support for the reliability of REO trace methodology.

Zhang et al. (2001, 2003, 2009) described a tag and detection method for REO tracing in an agricultural scour, erosion and infiltration settings which has been used by agricultural soil scientists (e.g. Deasy and Quinton, 2010; Polyakov and Nearing, 2004) to trace fine sediment particle movement. This published research discusses the effective functionality (minimal signature material loss and effective tracer mass balance and transport monitoring) and provides a basis upon which to test, refine and robustly adopt a similar methodology appropriate for application to SuDS. It is this method which has been modified (Sections 3.6.6 – 3.6.8) in the present thesis to fit the SuDS application, dictating that urban surface release, multiple event tracing and source to sink analysis of fine sediment mass balance analysis are traceable.

The sediment carrier used for tracing can either, be imported material (i.e. quarried fine gravel, washed sand and kaolin clay) artificially mixed to simulate urban PSDs or, be the actual source material taken from the urban surface (i.e. exact PSD of the car park, road or roof). In this thesis, preference was given to imported material for two reasons: (1) using clean (washed) material limited the potential for pH or humic matter influence on REO attachment to the sediment (Section 2.8) or possible contamination of the receiving stormwater flow path by heavy metals and minerals from the introduced tagged sediment; (2) the mass and PSD of sediment of the field source samples could be easily replicated (in accordance with BS 1198, 119, 1200: 1976), without compromise to the study or environment. Thus, urban sediment characterisation (Section 4.2) from each source and site were used to inform the PSD generated and Section 2.2 evidence was used to determine a representative mass for sediment build on the urban surface (i.e. average 1 month dry period). This raw material was thoroughly mixed prior REO tagging.

Commercial REO powder tested and certified as 99.9% pure was sourced from the Ganzhou Hong De New Technology Development Limited Company, China. Of the initial 15 REOs (Section 3.3), three were not viable for use in the present project:

Europium (Eu) is difficult to analyse as it interacts with Barium oxide (BaO) during the mass spectrometry analysis leading to erroneous results; Lutetium oxide (Lu_2O_3) and scandium oxide (Sc_2O_3) costs are an order of magnitude higher than other REOs. The remaining 12 REOs were considered appropriate and pilot field tests were used to determine the ratio selection of trace:sediment (Sections 3.6.3 - 3.6.5), subsequent to the following 5-step procedure for sediment tagging:

1. Sediment to be tagged was air dried and separated (sieved) into three categories (<63un, 63um-250um, 250um - 2mm) as described for best chemical binding in Kimoto et al. 2006 (Polyakov et al. 2009). Appropriate quantities of each of these size groups were used to create the total REO tagged sediment mass for each source location release.
2. Soil was moistened to achieve 15-20% water content (Zhang et al 2001) using oxidised/distilled water. Water was added to the sediment during continuous mixing (in a concrete mixer) to ensure a consistent and even water content was achieved.
3. The selected REO trace material was added to the moistened soil in a dilute/solute form (Zhang et al 2001). The REO powder solution was added to the sediment while continued mixing occurred to ensure an even and consistent concentration of tagging (Polyakov et al. 2009). The REO/sediment mixture was mixed in a commercial mixer on low speed for 5 minutes, appropriate to achieving homogeneity of REO and sediment mixing (Zhang et al 2001, Polyakov et al 2009).
4. Tagged sediment was then left to rest, mixed, for 24 hours in a covered container (Kimoto et al. 2006, Zhang et al 2001) to allow for full REO absorption to sediment particles.
5. After thorough mixing, the tagged material was released to the surface or water environment.

3.6.3 Method of release

Three methods were considered for application of REO onto the urban surface: (i) direct application of REO solution (suspension in deionised water) by backpack pump sprayer

(Deasy and Quinton 2010); (ii) sieved deposition to a specified deposition depth (Matisoff et al. 2001); (iii) un-absorbed tracer mixing into existing soils to a specified depth (Zhang et al. 2003, Kimoto et al 2006). As the surface is impermeable, method (iii) was not viable. Method (i) was also deselected, given the high probability that cohesion effects of kaolin clays in a PSD solution would increase aggregation (clumping) and reduce spatial homogeneity of distribution. Thus, the alternative controlled sieve release method (ii) was adopted.

Tagged sediment was mixed to the predetermined PSD on site immediately prior to use and representative loading rates are listed in Table 3.4 (informed by Section 2.2). Each urban surface was discretised into a $1m^2$ grid. Above the centre point of each grid cell, a manual sieve of 2.5mm perforations was held 1m above the urban surface and shaken to distribute the prescribed loading. This process was repeated for each grid cell appropriate to achieving the total urban loading rates of Table 3.4 (Section 2.2 and Section 4.2.1) and distributing homogeneously across the runoff surface (similar method was used Polyakov et al. (2009)).

Table 3.4 Summary of REO tagged sediment loading rates (Section 2.2) and further tested in across the field sites for confirmation (Section 4.2.1).

Release surface	Sediment loading rate ($ton/km^2/yr$)	Sediment Mass (kg/ release)	Representative area of site to be tagged (release area) (m^2)	Ratio of fine sand to Kaolin tagged (based on PSD for the surface)
Campus swale field site				
Road	50	10	500	7:1
J4M8 field site				
Car park	36	21	7,732	7:1
Road	50	6	1,585	7:1
Roof*	7	5	9,464	7:1
Newcastle Great Park field site				
High intensity urban area	15	15	NA – Release from stormwater culvert^	2:1
Construction area	386			

Note: Sediment loading rates are taken from the literature review (Section 2.2) and further tested in across the field sites for confirmation (Section 4.2.1). *The sieved method could not be used on the roof surface due to health and safety constraints; instead, tagged sediment was released en mass into the down pipe conveying roof runoff directly to the SuDS network. ^Where tagged sediment was released into a manhole or piped culvert outlet, it was diluted a further 50% prior to release in order that clumping or artificial aggregation was minimised.

3.6.4 The sample analysis

REO tracer analysis (i.e. concentration correlations of REO species, grain size and heavy metal/mineral analysis) requires acid digestion and mass spectrometer analysis to quantify the trace concentration in each sample collected. The REO trace analysis methodology follows standard Inductively Coupled Plasma Mass Spectrometry (ICPMS) described in the EPA Method 3005 and used by Kayhanian et al (2012) and Zhang et al. (2001, 2002, 2009). REO trace analysis methodology is detailed in Appendix III.

3.6.5 ICP-MS analysis

Whilst a full review of ICP-MS techniques can be found in (Jenner et al. 1990, Wolf 2005), a brief summary appropriate to the present thesis is as follows: ICP-MS uses argon gas plasma to quantitatively measure trace (REO) element concentrations in solid or liquid sample material. Strongly-charged ions (resulting from plasma molecular disassociation) are aimed at a mass spectrometer. The ionised elements of concern are filtered through multiple lenses through to the mass spectrometer, providing an identifying signature specific to the sample (illustrated in Figure 3.10). The electron multiplier dynode receives the filtered ions and acts as a detector, calculating the concentration of the element in the tested sample. The results are provided in parts per million (ppm) or parts per billion (ppb), as applicable.

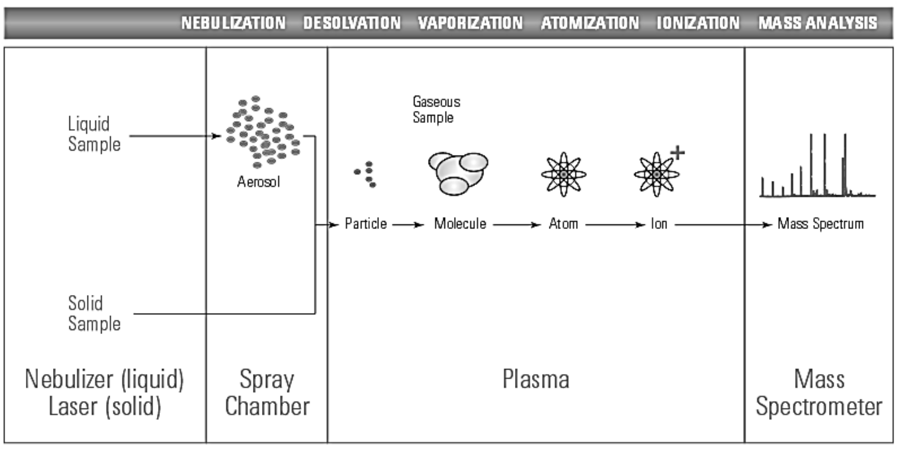


Figure 3.10 Schematic of ICP-MS sample disassociation (from Agilent Technologies 2005, Figure 1)

The ICP-MS used to analyse the REO concentrations from field samples was an *Agilent 7500ce*, maintained and provided by the Scottish Universities Environmental Research Centre (SUERC). This system is state-of-the-art, using the octopole reaction system (highly sensitive sequential ion filter based on the mass:charge ratio) via CeO^+/Ce^+ ratios (mass:charge ratio) of approximately 1% (Agilent Technologies 2008, Wilbur and Soffey 2004). The elements and their respective parameters relevant to the REO ICP-MS analysis are presented in Table 3.5.

Table 3.5 REO elements (x12) and ICP-MS analysis parameters as relevant to the REO methodology used in the present thesis.

Element	Analytical mass	Limits of detection		Variance or errors in analysis (RSD%)
		minimum (part per trillion, ppt)	maximum (parts per billion, ppb)	
Ho	^{165}Ho	0.02	100	0.56-1.46%
Pr	^{141}Pr	0.008	100	0.69-1.16%
Sm	^{147}Sm	0.07	100	0.7-1.72%
Nd	^{145}Nd	0.03	100	0.77-1.1%
Gd	^{157}Gd	0.03	100	0.89-1.23%
Yb	^{173}Yb	0.06	100	0.86-1.43%
Er	^{167}Er	0.08	100	0.44-1.11%
Dy	^{163}Dy	0.09	100	0.39-1.22%
Tb	^{159}Tb	0.01	100	0.38-1.45%
Y	^{89}Y	0.01	100	0.61-1.41%
La	^{139}La	0.01	100	0.77-1.31%
Ce	^{140}Ce	0.01	100	0.63-1.16%

3.6.6 Selection of REO trace:sediment ratio

There is limited guidance on the effective concentration of REO trace to sediment ratio, with agricultural field tests using up to 500g/kg of REO trace to ensure a clear trace signature in natural diffuse-pollutant environments (Deasy and Quinton 2010; Section 2.8.4) and 5-100g/kg in more controlled flow systems (Zhang et al. 2003, Deasy and Quinton 2010, Kimoto et al. 2006). As the nature of a trace is to provide detailed sediment transport information without significant influence to the receiving environment or sediment dynamics it is important to identify the minimal concentration of sediment trace necessary, yet without compromise to the results due to weak signature strength. Pilot studies were, therefore, carried out at the Campus swale site (Section 3.2) to identify the minimum effective trace concentration necessary for blue-green infrastructure sediment transport tracing. A key hypothesis of these tests was that the constrained flow path of SuDS would ensure tracer retrieval concentrations higher than that of the diffuse tracer studies in agricultural scenarios (Deasy and Quinton 2010), such that release concentrations far less than 500g/kg would be required. Thus, tagged sediment of a low (10g/kg) and moderate (100g/kg) trace to sediment ratio (i.e. in line with Zhang et al. 2003) were tested concurrently. Motivation for this selection was augmented by the swale directly discharging to receiving waters, such that use of lower tracer concentrations posed less environmental risk.

These pilot experiments compared the use of two unique rare earths (La and Nd) at different trace concentrations (La:10g/kg and Nd:100g/kg respectively). As both tracers were adsorbed onto the same imported sediment (Section 3.6.2) transport processes were reasonably assumed equivalent such that direct comparison of tracer performance could be assessed. Crucially, background concentrations of these REOs in both artificial runoff and swale soil were below ppm analysis levels, ensuring that analysis focussed solely on artificial tracer movement. Details of the field sampling locations, artificial flows and tagged sediment release location have been provided in Section 3.3.1. Hence, ICP-MS results are presented below to justify the selection of REO trace : sediment ratios, as used within the main programme of tracer field experiments in this thesis (J4M8 and NGP; Chapter 5).

Campus swale field test results

Figure 3.11 presents the ICPMS analysis for La (10g/kg trace concentration) and Nd (100g/kg trace concentration) field tests in the Campus swale.

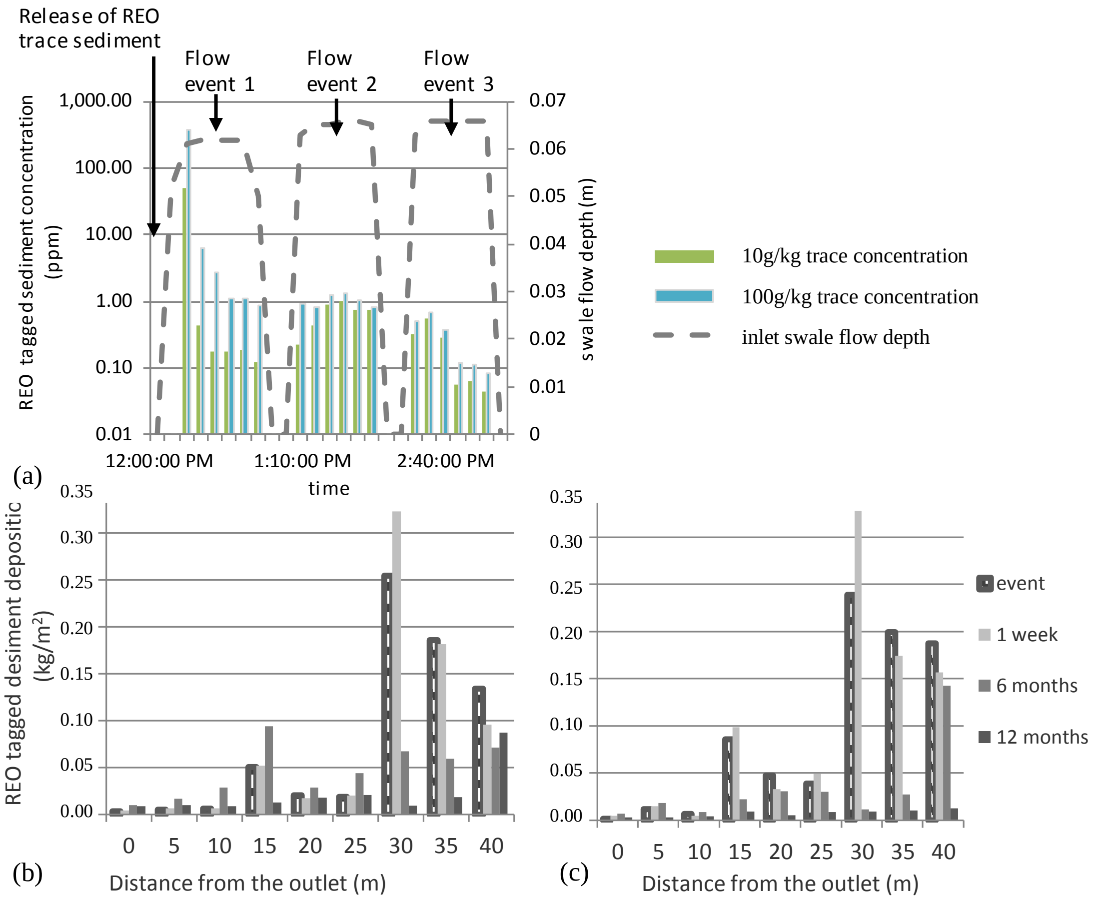


Figure 3.11 Moderate (blue) and low (green) REO tagged suspended sediment concentrations in the swale comparison to establish the appropriate tracer: sediment ratio use in urban vegetated stormwater drainage field experiments. Figure 3.11(a) presents the suspended sediment REO tagged sediment concentrations over the three artificial flow events. Figure 3.11(b) and (c) present the deposited (core sample) REO tagged sediment over the following 12 months for the two REO tag: sediment ratios. This analysis provides field data that supporting comparison and selection of an effective REO trace: sediment ratio.

Four important findings are drawn from Figure 3.8 data (also presented in Allen et al. 2015):

1. The general tends of REO tracers are similar, independent of tagged concentration used.
2. For runoff event 1, the concentration of 100 g/kg tagged sediment is an order of magnitude greater than the 10 g/kg tagged material. This appears a facet of the absorption maxima for the tagged soil composition being exceeded for the 100g/kg trace. To elucidate, laboratory analysis undertaken by Kreider (2012)

(b)

(c)

suggests clay/silt material adsorption maxima to be 12,400 ppm and a similar range (1900 to 43,000 mg/kg) is suggested by Spencer et al. (2007). Given that the 100 g/kg REO concentration is significantly above these adsorption levels, the excess REO Nd is released as solute. The lower concentration of 10g/kg was considered to be more appropriate to the investigations herein.

3. For events 2 and 3, the concentrations of 10g/kg and 100g/kg REO tagged sediment show similar (+/- 0.1 orders of magnitude) concentrations. This independence to tracer concentration for multiple event monitoring supports the use of the more conservative tracer concentration; this is particularly beneficial in view of its deployment into the natural environment.

4. Consecutive runoff events 1, 2, 3 exhibit progressively decreasing sediment tracer concentrations; this is also shown intra-event for runoff events 1 and 3. This reflects continual downstream conveyance and discharge of tagged sediment from the swale over the course of cumulative events. Thus, the REO method meets a key research objective (Section 2.2) of being able to visualise and quantify changes in tracer detention over time.

In conclusion, the lower concentration 10g/kg tracer was selected for further use in the field research of this thesis.

3.6.7 Multiple source (spatial) tracing

The SuDS assets within J4M8 (Figure 3.3.2) collect stormwater runoff from three urban sources: road, car park and roof surfaces. Criteria 4 in the trace requirements (Section 3.6.1) stated the need for the selected trace methodology to individually and simultaneously measure the movement of sediment from multiple sources through a single SuDS asset or network. Thus, tracer sediment was tagged (Sections 3.6.2 and 3.6.3) with unique REO signatures, specific to each different source (road, roof, car park) Whilst the detailed analysis of field monitored REO tagged sediment movement is provided in Chapter 5, a brief discussion is presented here to justify the method as fit-for-purpose.

Figure 3.12 provides an example dataset of REO tracer concentration along a single long swale in J4M8. In this example, with sampling undertaken near the inlet and outlet only. As the long swale conveys stormwater (and therefore potentially sediment pollution) from all three J4M8 urban surface sources, it is only via use of the unique REO identifiers per source that the land-use (source) can be visualised and analysed.

Use of the 10g/kg tracer concentration for three different REOs (Nd, Dy and Y) are proven able to measure and distinguish source-specific relationships and the methodology is therefore use in Chapter 5 more widely to examine pollutant delivery rates, in-SuDS transport rates, retention/detention rates, sink locations etc.

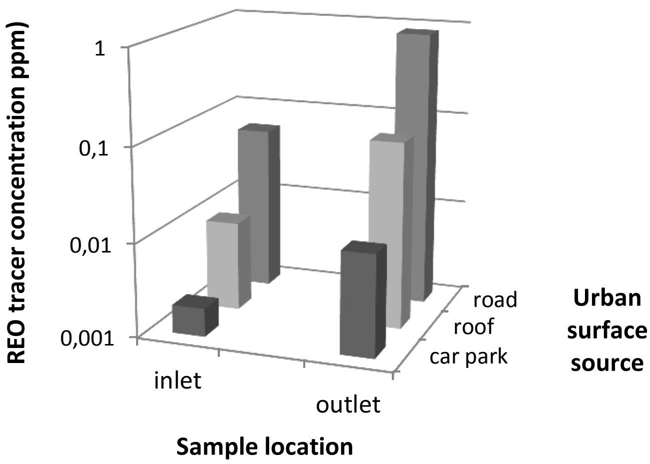


Figure 3.12 An illustration of source specific fine sediment visibility through one J4M8 SuDS asset (the linear wetland) for a single sampling timeframe. These data show that car park sources provide a lower REO concentration to the swale than material from other sources and illustrate spatial trends within the asset (detailed analysis and interpretation of source-sink relationships and asset performance is presented in Chapter 5).

3.6.8 Time-stamped release (temporal) tracing

Monitoring of source specific fine sediment movement through SuDS requires temporal tracing of urban sediment movement. Monitoring of tagged urban sediment pollution extended over 12 months at the J4M8 field site, and 6 months at the NGP site. Scientific rigor requires that experiments are repeated to ensure that experimental results are not localised anomalies or unrepresentative of the environment or processes being tested. Thus, the present methodology included releases of REO tagged sediment which were repeated at 4-12 month intervals (see Figure 3.13) on every urban surface source under investigation. Each timestamp had its own unique REO, as well as being unique to the source typology. The REO tracer, release timing and release location are noted within Figure 3.13 to help define where and when the release-monitoring-analysis activities were repeated in the field.

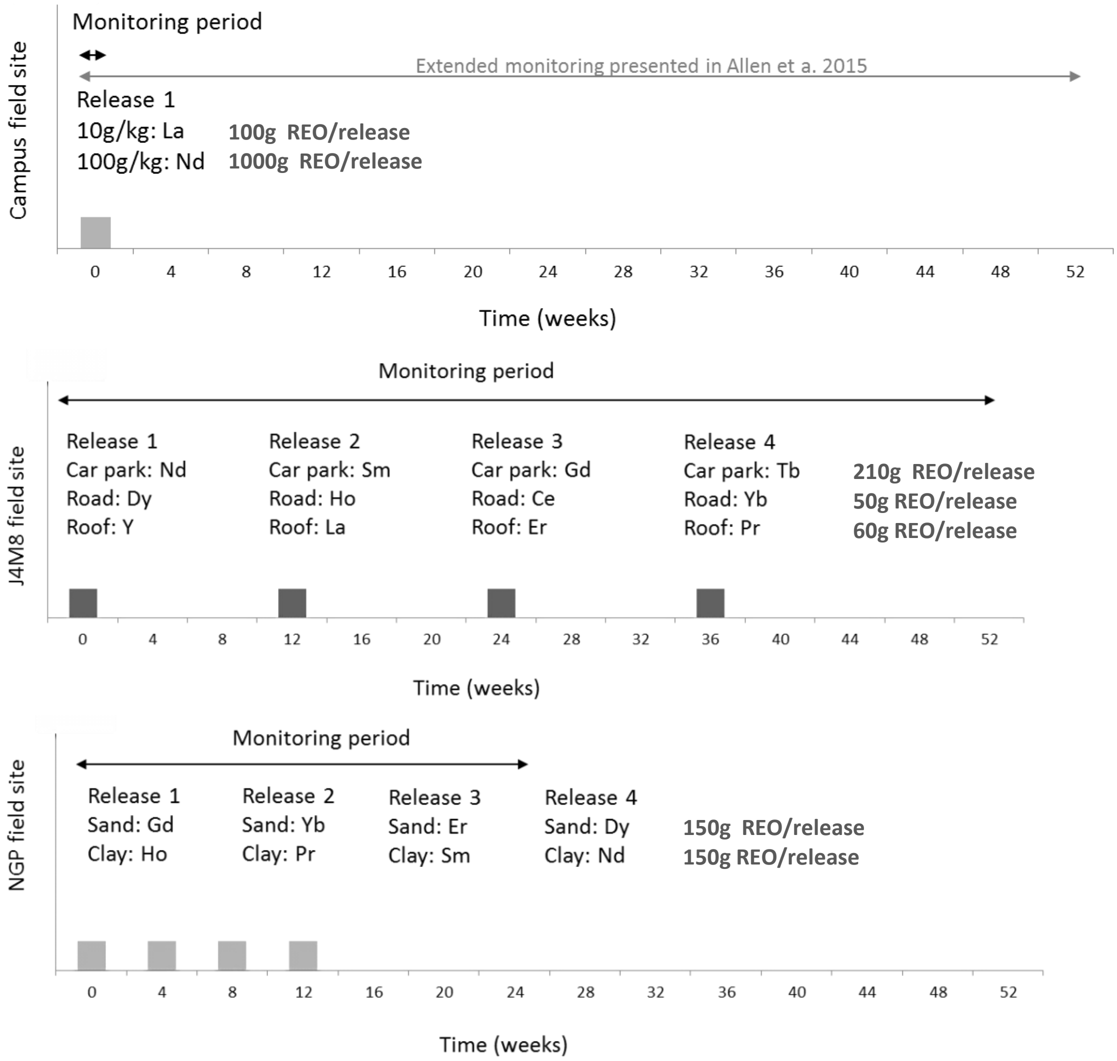


Figure 3.13 Sketch of the release time, location and monitoring period for each of the field sites included in this thesis research (Section 3.2).

The Campus swale field site (Section 3.3.1) was used to examine whether REO tracers and monitoring methodology worked effectively over extended periods up to 12 months. The time-stamped REO release of the first artificial runoff event (Section 3.6.6) yielded the REO tracer source for subsequent runoff-transport-deposition cycles. Bed deposition and core monitoring (Section 3.4.1) was completed between runoff events 2 and 3 and then repeatedly over the subsequent 12 months, such that tagged sediment deposition analysis is presented in Figure 3.14.

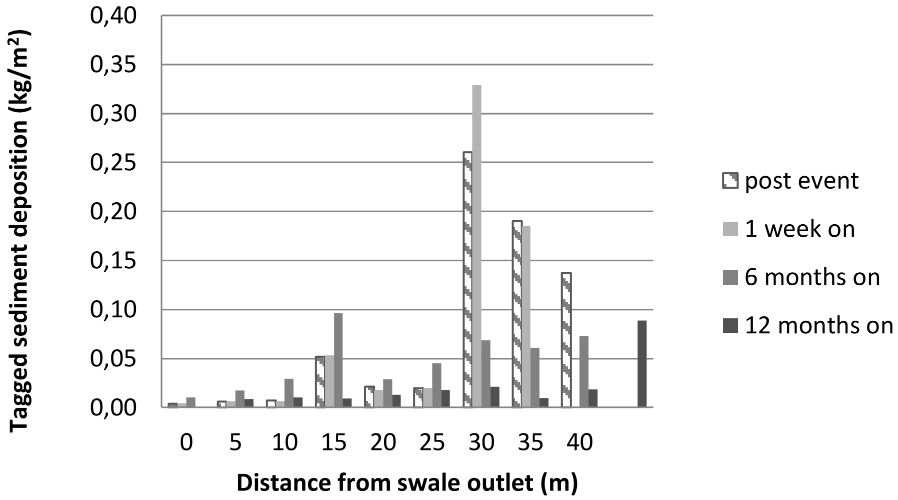


Figure 3.14 An illustration of the fine sediment concentration flux (within core samples) over an extended time period, from a single release and single urban source (road) within a single SuDS asset (Campus swale). Core samples were collected immediately after the artificial flow events (post runoff event 3), +1 week, +6 months and +12 months (La REO tracer results, 10g/kg trace: sediment ratio).

Figure 3.14 indicates that a specific REO signature can be discretely evaluated using ICP-MS, as essential to tracing a time-stamped sediment release in measureable concentrations and quantities over the medium-long term (12 months). The periodic sampling interval does show natural variability in the concentration over the 12 month timeframe, due to differences in sediment carrier characteristics, associate particle step-lengths, local deposition-erosion cycles, local resistance/turbulence etc. (Allen et al. 2015, Chapter 5).

3.6.9 Particle size specific tracing

The urban sediment particle size distribution (Section 2.2) was used to undertake an educated assessment of relative size potential effects on REO concentration (Section 3.3; Hubbard 2012, Timperley et al. 2005, Selbig et al. 2013), the REO fine sediment trace methodology was extended to consider size specific (i.e. fractional) transport within a SuDS asset. As fine sediment <62 µm has a higher surface to volume ratio, slower particle fall speed and longer period of time to settle than larger sediment (62 µm – 2mm) (Van Rijn 1984), using this dimension to threshold the clay fraction from silts/sands permits consideration that clays can adsorb higher concentrations of REO

(Deasy and Quinton 2010, Zhang et al. 2003, Polyakov and Nearing 2004, Kimoto et al. 2006).

Trials were undertaken at the Newcastle Great Park (NGP) SuDS pond field site (Section 3.3.3). Released sediment into the pond inlet comprised two known sizes (kaolin clay 2-5 µm and fine sand 150 µm) with fraction-specific tracers generated via methodology of Section 3.63. Tagged fractions were not mixed prior to release (to avoid potential enrichment via transference, Kimoto et al. 2006), but were simultaneously released at exactly the same inlet location. Four releases were completed at monthly intervals, each with time-stamped unique identifiers. In total 8 REOs were required for these tests (2 fractions x 4 releases)

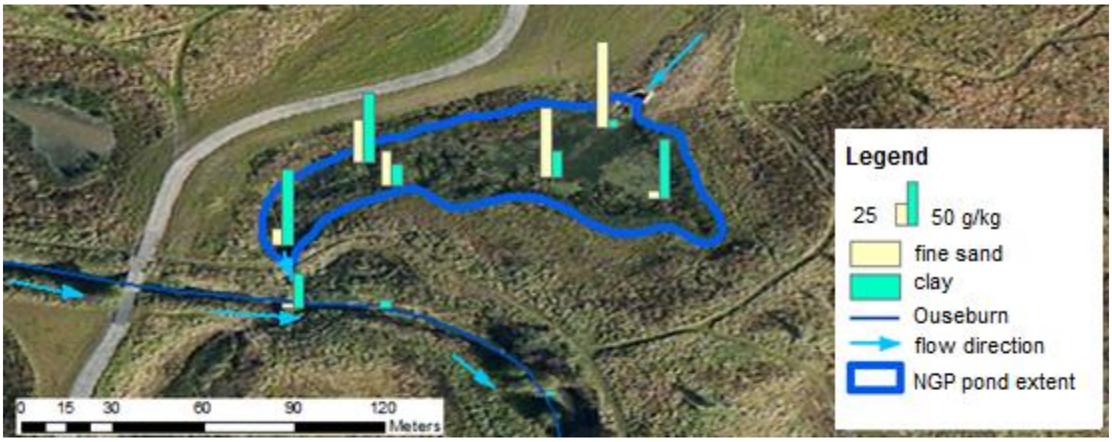


Figure 3.15 Example of possible particle size deposition distributions within the six sediment deposit monitoring sites across the NGP field site SuDS pond, as delineated by REO trace monitoring The bar charts represent sand (yellow) and clay (green) REO tagged sediment deposition (g/kg per sample location) with the greatest sample result with a reference range of 25(yellow) and 50(green) presented in the Legend.

Figure 3.15 illustrates an example of REO trace sediment deposition distribution for an example sample period. Using unique tracers for the two separate particle sizes (sand and clay) allows the fine sediment transport and detention to be characterised by particle size. Figure 3.15 identifies the quantity of sand tagged sediment deposited within each sample location to be different to tagged clay deposition. The deposition and transport of sand material can thus be compared to clay material by using particle size specific REO tracers. When combined with heavy metal pollution analysis, which is influenced by sediment particle size, particle size specific REO tagged fine sediment transport provides a more detailed illustration of sediment pollution movement through a SuDS asset.

In summary, the REO method, specific to the 12 elements (listed in Table 3.4) suited to ICP-MS, has been tested for the following uses: (i) single release; (ii) multiple source identifiers; (iii) time-stamped identifiers and (iv) particle size specific identifiers. It has been successful in providing measurable REO concentrations in solute and sediment samples over spatial scales m-km from source and over temporal scales up to 12 months. This supports their use in the longer-term investigation of SuDS source-sink tracing for specific particle size asset performance analysis. However, as with all tracer studies, it is prudent to recognise the limitations and uncertainties of the methodology proposed and justify why these can be tolerated for the present study (Sections 3.6.10-3.6.12).

3.6.10 Potential methodological errors due to invasive sampling

As the mass balance of tagged sediment is discussed in Chapters 4 and 5, it is crucial to acknowledge that the monitoring and sampling process in the field work results is some loss of sediment material from the SuDS network. The losses due to invasive sampling (cores and deposit extraction) are presented in Table 3.6 indicating that the quantity removed due to sampling is less than 4% in all cases and, more typically, less than 1%. The error due to influence of material loss from sampling activities is therefore very small when reviewing data presented in later Chapters of this thesis.

With regards to total sediment transport, field samples were 0.5L (solute) in volume and $0.005m^3$ (sediment samples). The total sediment mass in suspension in the SuDS network ranges from 0.01-over 1000 mg/L (Section 3.4.1), with a deposition load of 0.01-30 kg/m^2 (Section 3.4.1; detailed mass deposition analysis presented in Chapter 4). The quantity of sediment removed from the SuDS network was asset specific, with linear wetland (J4M8) and the pond (NGP) showing the greatest mass of sediment removed. When the total volume of sediment in suspension and deposited on the bed is estimated, surface and bed deposition sample losses also result in less than 1% of the total sediment in the SuDS system.

Table 3.6 Percentage of REO trace removed from the SuDS network due to sampling activities relative to the location of tagged sediment release. The J4M8 site (Section 3.3.2) provides road, car park and roof data, whilst the NGP site (Section 3.3.3) reflects urban development surface losses.

	Network 1: Car park REO trace	Network 2: Roof REO trace	Network 3: Road REO trace	Urban development REO trace (NGP)	Experimental monitoring period
Release 1 (total)	0.549%	3.174%	0.69%	0.598%*	52 weeks
Release 2 (total)	0.001%	1.803%	0.11%	0.489%*	40 weeks
Release 3 (total)	0.001%	1.302%	0.05%	0.952%*	28 weeks
Release 4 (total)	0.0002%	0.132%	0.00%	0.458%*	16 weeks
Over the first 4 weeks					
Release 1	0.002%	0.236%	0.07%	0.145%	4 weeks
Release 2	0.123%	0.173%	0.04%	0.164%	4 weeks
Release 3	0.226%	0.292%	0.01%	0.466%	4 weeks
Release 4	0.058%	0.020%	0.00%	0.249%	4 weeks
Average	*0.102%*	*0.180%*	*0.03%*	*0.293%*	*4 weeks*
Standard Deviation	*0.08%*	*0.10%*	*0.03%*	*0.13%*	*4 weeks*

* The urban development REO trace results are for NGP field site. This site was monitored for 24 weeks rather than 52. The results with a (*) indicate the total experimental period REO loss due to sampling at this location for REO tagged sediment release 1 to 4.

3.6.11 Potential methodological errors from size-specific REO adsorption

There is some evidence that REOs may preferentially bind to fine particulate material such as silt and clay particles (Kimoto et al. 2006). Therefore, where a large particle size distribution, PSD, (including coarse sediment, sand or gravel) is used in a trace experiment, there may be an over or underestimation of REO concentration due to REO tracer transference. Research in REO tracer enrichment due particle size re-distribution during erosion experiments suggests a potential error of 4% when considering a particle size range from 8 mm to below 0.9 mm (Zhang et al. 2003, Deasy and Quinton 2010, Polyakov and Nearing 2004, Kimoto et al. 2006). However, as the present thesis restricts PSD to a far tighter size range focussed specifically on clays, silts and very fine sands the error is likely even smaller (see Section 3.6.9).

3.6.12 Potential methodological errors from enrichment

Firstly, enrichment from REO existing within the swale soil bed material is not considered significant for this site as background levels are below analytical detection limits. Previous REO tracer research (undertaken by Deasy and Quinton (2010), Zhang et al. (2003), Polyakov and Nearing (2004)) provided limited or no publication of analysis error due to soil REO enrichment. Kimoto et al. (2006) presented enrichment analysis as part of their sediment trace experimentation, demonstrating a possible error in results of up to 4% due to enrichment when considering particle sizes between 0.9-8.0mm.

Secondly, enrichment due to change in environmental conditions, such as acid rain, change in runoff flow or soil pH, is also considered insignificant for these field sites. The field sites are located within urban developed areas and originally constructed from ‘clean’ fill (DEFRA 2009) on greenfield sites without land remediation requirement. Artificial runoff events employed across the Campus swale used potable water tending to a neutral pH (7.0); with average rainwater pH 5.6 the difference is not considered significant enough to cause REO concentration error (Kimoto et al. 2006, Wen et al. 2013). All other field experiments were subject to natural rainfall occurrence in non-industrial regions without any evidence of acidification of rain (Eynon and Switzer, 1983).

Thirdly, enrichment due to REO transfer is considered the most likely of the three processes described here. It is acknowledged both, that elevated REO errors may occur when tagged material is a light textured soil (due to low aggregate forming abilities); and, that preferential bonding with small particles may cause non uniform binding if a fine-coarse mixture is employed. There is also the possibility for transference of REO trace material from a coarse to a fine particle during the runoff processes; for example, Kimoto et al. (2006) illustrate a greater REO concentration associated with particles <0.09mm and show changes in REO concentration of +/-4% due to active transference towards a smaller particle sizes; this order of magnitude of change is generally accepted in similar studies (Zhang et al. 2001, Polyakov and Nearing 2004). Thus, Section 3.6.2 and 3.6.3 show tagging and release methodologies specific to fractions and maintaining particle disaggregation; this should minimise errors from transference to less than that stated in Kimoto et al.’s (2006) research.

3.7 Summary of experimental methodology

Three field sites, containing established SuDS assets, have been identified and monitored over a period of 6 to 12 months to provide a multiple rainfall-runoff event dataset comprised of time-stamped rainfall, flow, sediment and pollutant characteristics (data). This dataset has been complied through sampling and analysis based on both standard and novel techniques discussed in this Chapter. In summary, the field collection and sample analysis methodology incorporated:

- A review of current fine sediment tag and trace methodology to identify options appropriate to the field sites and research objectives;
- Pilot testing of a novel fine sediment pollution tracing methodology to examine its effectiveness in tracing multiple event SuDS fine sediment movement;
- Existing best practice field sampling methods (collection of surface and deposited sediment pollution material);
- Current British Standard methods for standard analysis (i.e. TSS, deposition and heavy metal analysis); and,
- Description of the novel REO fine sediment trace methodology designed for implementation across ephemeral and perennial vegetated SuDS over an extended (+6months) monitoring period.

4 Mass sediment movement and deposition within SuDS assets and networks

4.1 Introduction

Urbanisation and the intensification of urban development are key sources of sediment pollution in local watercourses. The EU Water Directive, Environmental Quality Standards Directive and UK planning policy guidance (SEPA 2010, Scottish Government 2014, Pickles 2014, Department for Communities and Local Government 2012) acknowledge this and provide guidance towards achieving acceptable water quality levels. Sustainable urban Drainage Systems (SuDS) are the preferred 'naturalised' stormwater treatment method to combat urban runoff pollution and have been increasingly implemented in Scotland over the last 20 years.

CIRIA guidance states that SuDS should be designed to effectively mitigate pollution from urban and developed land uses (CIRIA 2007). The specific water quality expectations, as a percentage reduction or acceptable water quality standard, are not clearly stated in either the current (2007) or new (2015) SuDS manuals. Instead, there is an assumption that SuDS function to detain sediment washed off the urban surfaces. This detention is via increased hydraulic residency time within an attenuation/storage asset and/or high surface roughness within the flow path (i.e. use of vegetation); both reduce local flow velocities, encouraging sedimentation (CIRIA 2007). Past research has focused on event based stormwater quality and quantity achievements of SuDS (Chapter 2, Table 2.7), with limited field monitoring or detailed analysis of long term efficiencies. The assumption is that once pollution has entered the SuDS system, it will be "*contained within the [upper] SuDS components so minimising the damage to the drainage system, and helping ensure the high concentrations of contaminants are not conveyed to the receiving watercourse*." (CIRIA C697 2007, pp 3-12). Without actual data on long term, multiple rainfall-runoff event relationships to stormwater treatment efficiency, validation of this assumption, particularly with regard to sediment, is not possible. Within this context, the research described in this Chapter presents a fortnightly time-stepped sediment flux dataset for range of SuDS assets and networks over multiple rainfall-runoff events. Data consider long term timeframes (≥6 months) to quantify water quality treatment efficiency of both individual assets and the combined network performance, in two case study sites (J4M8 and NGP).

4.2 Sediment supply & associated empirical estimation

Before considering how effective SuDS networks are for water quality improvement, the quantity and quality of the influent, and its source, need to be identified. The standard mass balance equation rests on the knowledge of both inflow and outflow contaminant concentration and mass:

$$C_{in} = C_{lost} + C_{detained} + C_{enriched} + C_{degredatded} + C_{out} \qquad \text{Eqn.11}$$

where C represents the pollutant, in this case fine sediment, concentration or mass. The literature (Figure 2.3; e.g. Zanders 2005, Van Metre and Mahler 2003, Hubbard 2012) illustrates urban sediment characteristics and loading that can be used to estimate $C_{in.}$ Field sampling and analysis, across the J4M8 and NGP field sites, has been conducted to identify whether the selected field sites are representative of general urban surfaces (with regards to pollutant loading) and to provide site specific urban pollutant loading context for the field site monitoring.

4.2.1 Characterisation of fine urban sediment

Surface loading

The surface collections of source sediment (Section 3.4.2) were used to identify the site specific mass and particle size distribution (PSD) characteristics of the urban sediment. In all cases, the samples were taken from a $1m^2$ area of the car park and road surfaces, during a dry period, prior to REO tagged sediment release. Data are presented in Figure 4.1 and compared to the literature review (Chapter 2, Figure 2.1).

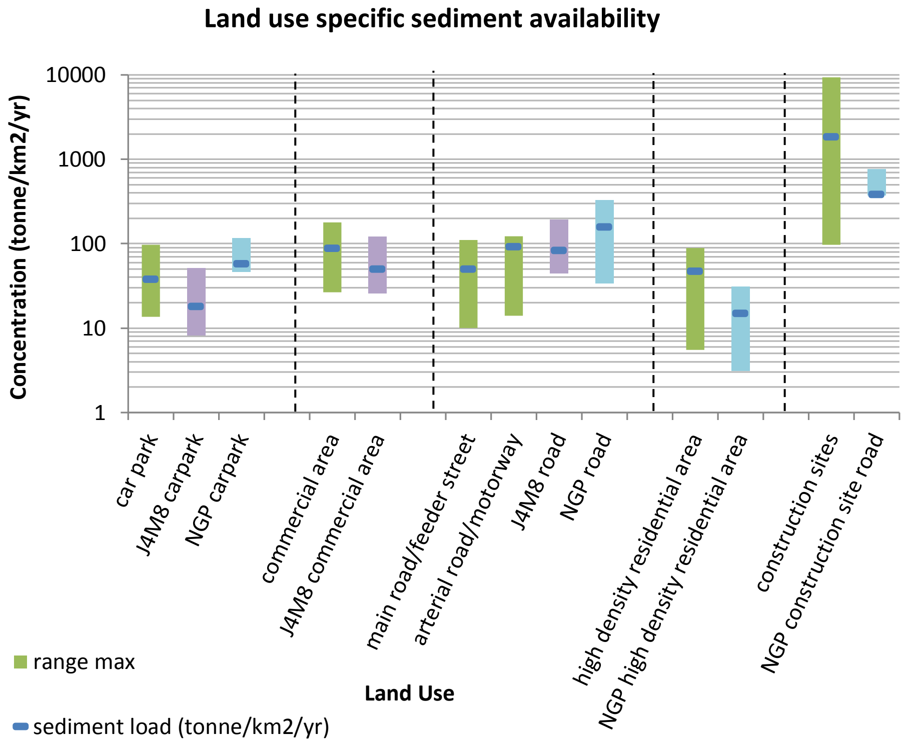


Figure 4.1 Urban sediment loading (surface deposition) from field sites. Literature review values are illustrated in green, J4M8 field results are purple while NGP results are blue. The average values are presented as **dark blue points (-)**. The urban surface load indicates the available material that could potentially be washed off during a rainfall-runoff event. The proportion of sediment moved by rainfall-runoff events is dependent on the event characteristics as well as urban sediment characteristics (e.g. particle size distribution) (Ellis 1986).

The surface area annual loadings found within J4M8 and NGP field sites generally reflect the range found in published literature (Figure 4.1). The NGP car park and road surface sediment availability show an average and range maxima notably more elevated (average by +60 tonnes/km^2/yr) than previous data for main or arterial road surfaces. This appears a direct response to redistribution of local construction site sediment. Although Figure 4.1 shows that best practices at the construction site mitigate successfully the impact of direct sediment availability on immediate construction roads (to mid-range levels of the literature data), the maximum availability of material on these local roads (up to >300 tonnes/km^2/yr) closer reflects that of the construction roads (>300 tonnes/km^2/yr) and indicates current higher vehicle use of the NGP roads by construction traffic.

The NGP high density residential sediment availability minima (data) fall outside the range of previous studies. For NGP, this may just reflect the stronger blue-green design of the urban space, where high density housing has greater green space and infrastructure built into the plot cartilage (e.g. paths, drives). The J4M8 roads also show slightly higher maxima availability than the literature; as this site is not affected by construction, when considered alongside the similar NGP data, it may reflect progressive increases in vehicle abundance and usage over the last decade (George and Kershaw 2016).

The field sampling of urban surface loading across both field sites confirms that the sediment loading published in the literature is generally indicative of this thesis' field conditions and the REO tagged sediment can robustly be considered representative of urban (UK) sediment loading rates; this justifies the methodology of Section 3.5, provides confidence in trend analysis and supports upscaling of conclusions across UK urban areas.

Particle Size Distribution (PSD)

Sediment Particle Size Distribution (PSD) differs according to the urban surface (Figure 2.2) with generally accepted d_{50} dimensions for sediment sourced from roof, road and car park to be, respectively, 102µm, 70µm and 28µm respectively. Section 2.2 highlights the importance of understanding the PSD, source and hydraulic residency of $\leq$ 2mm grains (Section 2.2), in order that estimates of sediment (re)entrainment and transport through the SuDS network can be developed, as required for sediment detention efficiency calculations.

For NGP and J4M8, all urban surfaces were tested for PSD using the methodology of Section 3.5.1 (*Mastersizer S* laser analysis). Figure 4.2 illustrates the results specific to the sample location (car park, roof or road location).

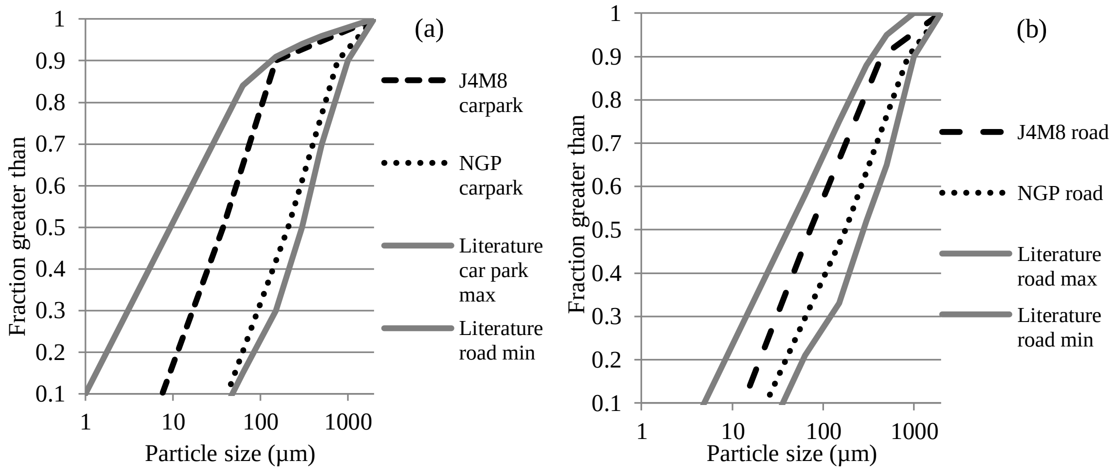


Figure 4.2 Particle size distribution of urban sediment for the (a) car park surface and (b) road surface sediment. Published urban sediment PSD, summarised in Figure 2.2 (Chapter 2) illustrates >80% of urban material to be <2mm, thus supporting Figure 4.2 bounds.

Field survey PSD data for both the J4M8 and NGP field sites comfortably fall within the overall range of literature review PSDs. Two notable findings are drawn from Figure 4.2: Firstly, the J4M8 material is finer than NGP, independent of source location. In explanation, as part of the NGP construction operations, dust management measures have been implemented. These include water spraying (to prevent construction dust becoming/remaining airborne) and street sweeping. Construction dust management measures are generally focused on fine sediment (CIS36rev2 (2013), Kukadia et al. 2003). As a result, the NGP sediment illustrate elevated d_{50} (150-200μm) and d_{90} (≥1mm) results compared to J4M8, with the finer particulate material being cleaned-off or washed off the urban surface adjacent to construction areas as a result of construction dust management activities.

4.2.2 Urban sediment movement into SuDS networks

Urban source provision of fine sediment is the key limiting and controlling factor with regards to urban runoff pollution (Vase and Chiew, 2003; Bai and Li, 2013). Once the urban sediment characteristics and loading rates are known, an estimation of the wash-off rate can be made based on a set of complex environmental factors including rainfall intensity, occurrence, material characteristics and availability, site and receiving waterway/drainage design. To determine the wash-off rate at from the J4M8 car park, roof and road surfaces, the quantity of REO tagged sediment (Section 3.6) remaining on the urban surface after release was monitored every month. The quantity of REO tagged sediment released was designed as equivalent to 1/12th of the annual sediment load for the surface type and area (no seasonality included in this assumption due to lack of detail available on seasonality influence on urban sediment loading). Over the 12 month monitoring period, samples were taken monthly to quantify the REO tagged material remaining on the road, roof and car park surfaces (Figure 4.3).

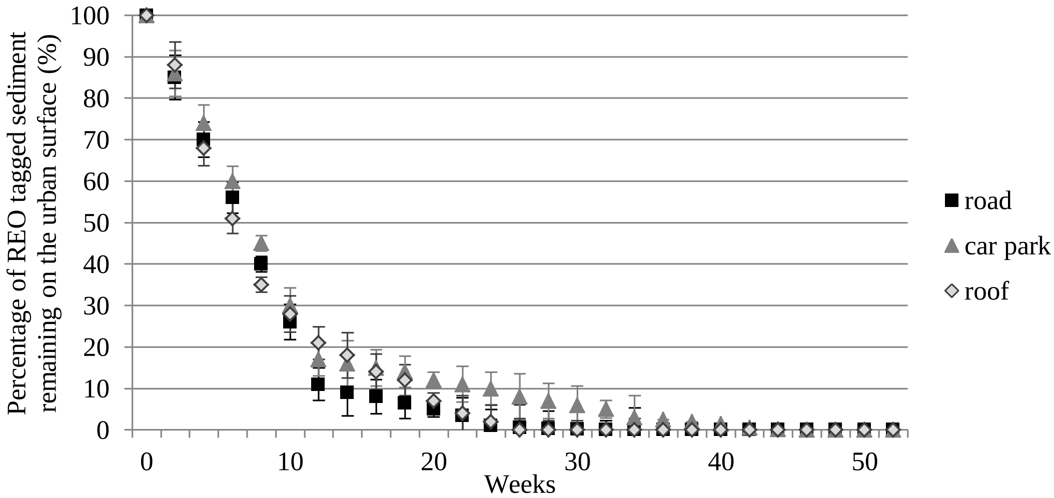


Figure 4.3: Monitored rates of change (%) in tagged sediment remaining on urban surfaces influencing the case study SuDS networks. Data is shown in % of original REO retained on the urban surface; the inverse of these data provides interpretation of wash-off rates.

For the first 2 months (8-10 weeks), all surfaces indicate similar wash-off rates with losses totalling 70-75% (i.e. approximately 30% remaining on the urban surface, Figure 4.3). The roof washoff shows a subtle difference to car park washoff, with a slightly higher initial rate of urbane REO tagged sediment release (car park and road error bars do not appear to overlap, suggesting a slight difference in trends during the first 2 months). Overall, trends in roof wash-off appear subtly distinct from other sources (tending towards exponential decay, rather than linear), suggesting more rapid surface flushing due to greater hydraulic efficiency of roof gradients and conduit conveyance in gutters. However, while the standard deviation range suggests the roof washoff trend may be distinct during the initial 2 months, after week 12 the distinction become less clear (error bar overlap) and the roof and road washoff rates become quite similar.

For the ground-level urban surfaces (road and car park) their rapid linear decay rate continues until week 12, when <20% of material remains on the urban surface. Subsequently, average wash-off rates reduce noticeably and begin to exhibit source-specific response. However, standard deviation response suggests road surface to be subtly distinct during weeks 12-20 and car park surface only to show distinction between weeks 20-32 (road and roof illustrating similar responses during this period).

Road wash-off is the faster process showing on average 95% removal after 6 months, equivalent to roof wash-off. Car parks took longer, up to 9 months to achieve equivalent wash-off amounts.

In explanation of the average trends found in Figure 4.3, the location of the surface within the urban context, adjacent SuDS construction and PSDs must be considered as combined influences. Firstly, both the car park and road flow paths are overland sheet flow, dictating shallower depth, smaller velocities and lower shear stresses for entrainment comparative (for equivalent storm events) to the roof water where piped flow systems contribute a more efficient wash-off role. Further, the length scales of sheet flow differs between the car park and road surface; for the road surface flow is constrained to and distributed across a limited road width, 7.5m, which is 1/5th of the car park width, 30-35m. As such, there is greater potential for the car park to exhibit convergent (rather than sheet) flow, which in turn increases the relative velocity, discharge and sediment wash-off across the surface.

A second consideration in interpretation of Figure 4.3 is the influence of the downstream boundary of the flow path on the sediment-wash off rate; i.e. whether it is an open ended pipe (above or below the standing water level) or a vegetated downstream boundary. Specifically, sediment sourced from the roof was conveyed via a stormwater pipe directly into the wetland; the water level of the wetland formed the downstream flow boundary and was designed to avoid backflow or surcharging. Conversely, the road and car park exhibit grass filter strip boundaries deliberately designed to increase in surface Mannings ‘n’ via vegetation resistance; this rapidly reduces local velocity to cause stormwater backflow up to the height of the grass (depending of vegetation flexibility) (Munoz-Carpena et al. 1999, Hogarth et al. 2003, Rose et al. 2003). This reduces shear stress to encourage local ponding and sediment deposition at the boundary between the car park / road surface and the grass filter strip (Figure 4.4). From site observations, it is reasonable that local distinction in the plant density and uniformity along the vegetation/impervious surface boundaries of the road and car park may have influence on wash-off rates. For the road, poor maintenance of the boundary evidenced flow short-cutting through the vegetated curb via scour channels; this would increase the rate of sediment wash-off from the road surface compared to that of the car park, which demonstrated a better-maintained, uniform vegetation boundary.

Figure 4.4: Photograph of sediment deposition due to backflow and ponding along the grass filter strip, taken at J4M8 09/05/2014. Photographed car park-filter strip boundary is adjacent to the area used in the field experiment.

4.3 Sediment movement within a SuDS network

Event specific and total monitoring period trends in sediment suspension and deposition within the SuDS network define the sediment transport efficiency of the network, and the assets therein. To analyse the multiple event transport and detention of fine sediment within a SuDS network it is important to first understand the general overall functionality of the network and its assets. The analysis presented in this Section is derived from work at the J4M8 and NGP field site.

4.3.1 Rainfall and flow characteristics for the sample period

The site specific monitoring provided continuous rainfall and flow monitoring (Section 3.4) within the SuDS network (within the wetland, linear wetland, swale (J4M8) and pond (NGP). Rainfall data is presented and analysed in Table 4.1 and 4.2 (for J4M8 and NGP respectively) below, whilst flow data is presented in Table 4.3. For the purposes of this study, rainfall events are defined as continuous rainfall resulting in a minimum of 0.2mm rainfall depth, with clustered rainfall events identified as occurring with no greater than 24hr ADD period between rainfall events.

Table 4.1 provides the fortnightly rainfall statistics relevant to this study. Average fortnightly rainfall total was 36.1mm and highly variable, up 98mm for specific events (St.Dev.30). The number of rainfall events within the fortnightly monitoring periods was also variable from 0 to 24, with each typically ~2 hour duration (average) but up to

23 hours for extreme events. Important to this thesis, the period of no rain prior to an event sample was up to 91 hours (3.75 days) and 21 hours (<1 day) on average; this means that wash-off samples may not fully reflect the actual ADD timeframes up to 13 days (average 8.5 days; St.Dev.3.4) due to sample frequency extracting material during the ADD period.

Table 4.1. J4M8 Rainfall data summary. The table specifically reviews the timeframes of events relating to the monitoring frequency, including making distinction between ADD (Antecedent Dry Days) and the dry period prior to sampling.

	Rainfall since last sample			**ADD during sample period**	**Dry period prior to sample**	**Most recent event**		
	Rainfall (mm)	Number of rainfall events	Average intensity (mm/hr)	hrs	hrs	Intensity (mm/hr)	Rainfall duration (hrs)	Rainfall (mm)
week 0	11.4	5	0.43	0	4	1.7	4	7
week 2	25	10	1.02	216	2	2.2	2	4.4
week 4	17.4	22	0.43	264	0	12	0.03	0.4
week 6	6.4	12	0.77	312	1.5	0.7	0.9	0.6
week 8	9.6	19	1.89	240	5	12	0.03	0.4
week 10	56.6	15	3.94	264	8	12	0.03	0.4
week 12	66.8	24	1.77	240	7.5	4.3	1.3	5.4
week 14	51.6	15	1.9	96	0	1.6	1.9	3
week 16	11.2	4	1.45	144	47	12	0.03	0.4
week 18	51.4	8	2.1	264	26.5	12	0.03	0.4
week 20	0.4	1	12	120	7.5	12	0.03	0.4
week 22	51.4	10	2.56	240	90.5	12	0.03	0.4
week 24	19	7	1.46	96	1	1.2	9	10.4
week 26	98	8	1.96	96	2.5	1.8	23	42
week 28	65.6	11	2.47	240	16.5	9.8	0.4	3.6
week 30	0	0	0	72	88.5	0	0	0
week 32	6.8	4	1.48	288	43.5	0.6	3	1.8
week 34	18	5	0.26	168	41.5	30	0.03	1
week 36	5.6	3	0.7	288	41.5	1.4	0.8	1.2
week 38	85.2	7	1.43	288	42.5	1.1	6	6.8
week 40	38.2	19	1.61	144	1.5	12	0.03	0.4
week 42	1.6	4	12	288	39	12	0.03	0.4
week 44	15	9	1.76	264	5.5	1.1	4	4.4
week 46	90.2	8	2.42	216	15	2.1	0.6	1.2
week 48	46.8	13	1.59	264	4	2.7	1	2.8
week 50	85.6	22	1.93	192	2	3	2	5.4
week 52	39.4	13	1.62	264	31	7.2	0.1	0.6
Average	36.1	10	2.3	206	21	6.7	2.2	4.0
Minimum	0.0	0	0.0	0	0	0.0	0.0	0.0
Maximum	98.0	24	12.0	312	91	30.0	23.0	42.0

Of particular note, Weeks 38 and 46 show the greatest rainfall totals over the two week period (90.2mm and 85.2mm respectively), coincides with only moderate event occurrence (10-12 individual rainfall events) yet higher rainfall intensity (up to 30mm/hr, compared to average statistics of 12mm/hr. Average rainfall intensity is calculated as the average of individual rainfall event intensities over the specified duration (e.g. fortnight, total monitoring period).

Table 4.2. NGP Rainfall data summary. The table specifically reviews the timeframes of events relating to the monitoring frequency, including making distinction between ADD (Antecedent Dry Days) and the dry period prior to sampling.

	Rainfall since last sample			**ADD during sample period**	**Dry period prior to sample**	**Most recent event**		
	Rainfall (mm)	Number of events	Average intensity (mm/hr)	hrs	hrs	Intensity (mm/hr)	Rainfall duration (hrs)	Rainfall (mm)
month 1	40	42	0.4	627	0	0.2	4	2.0
month 2	59.8	24	0.6	646	6	0.2	5	1.2
month 3	35.8	20	0.6	669	14	0.2	1	0.2
month 4	32	22	0.4	679	92	0.2	1	0.2
month 5	18.8	25	0.4	676	52	0.2	1	0.2
month 6	37.8	22	0.5	653	18	0.9	3	2.6
Average	37.4	26	0.5	658	30	0.3	3	1.1
Minimum	18.8	20	0.1	627	0	0.2	1	0.2
Maximum	59.8	42	2.25	679	92	0.9	5	2.6

Across the NGP field site, rainfall information (Table 4.2) was collected monthly from the nearby Bureau of Meteorology office rain gauge (Newcastle Airport; Section 3.4).

Rainfall ranged from 18.8-59.8mm per month, with an average monthly rainfall of 37.4mm. The greatest total rainfall occurred during month 2 of the NGP monitoring, while the greatest rainfall occurrence was during the first month of monitoring (month 1). Rainfall intensity is found to range between 0.2-0.9mm/hr and there is a direct positive correlation between total rainfall and greater rainfall intensity (with the exception of month 1).

The different sampling regimes employed across J4M8 (fortnightly) and NGP (monthly) result in un-comparable datasets presented in Tables 4.1 and 4.2. To allow direct rainfall/ADD characterisation comparison of these two field sites, monthly rainfall summary has been provided in Table 4.3.

Table 4.3. Monthly cumulative rainfall and ADD information for the J4M8 and NGP monitoring periods. The J4M8 field campaign occurred over the 12 months prior to NGP monitoring (a 6 month monitoring period in 2015). This table has been constructed as a comparative rainfall dataset for J4M8 and NGP rather than providing sample period specific data, as presented in Tables 4.1 and 4.2.

	J4M8 monitoring period (January - December 2014)				*NGP monitoring period (January-June 2015)*			
	cumulative rainfall (mm)	number of rainfall events	average intensity (mm/hr)	ADD (hr)	cumulative rainfall (mm)	number of rainfall events	average intensity (mm/hr)	ADD (hr)
month 1	54.6	39	0.3	653	40	42	0.4	627
month 2	14.4	27	0.1	659	59.8	24	0.6	646
month 3	135.6	43	0.7	687	35.8	20	0.6	669
month 4	103.2	26	0.7	691	32	22	0.4	679
month 5	124.2	22	2.0	691	18.8	25	0.4	676
month 6	114.2	16	0.9	662	37.8	22	0.5	653
month 7	25.8	10	0.5	665	No field sampling			
month 8	123.8	27	0.6	661				
month 9	16.2	12	0.2	735				
month 10	206.4	35	0.6	645				
month 11	63	24	0.7	674				
month 12	146.2	29	0.9	668				
Average	94.0	25.8	0.7	675	37.4	26	0.5	658
Minimum	14.4	10	0.1	645	18.8	20	0.1	627
Maximum	206.4	43	0.2	735	59.8	42	2.25	679

When rainfall and ADD periods for the two field sites are compared (Table 4.3), it is clear that a key distinction between sites is that J4M8 exhibits total rainfall double that of NGP, with more extreme rainfall intensities. There is less rainfall across the NGP field site, during the respective monitoring periods, compared to J4M8 (37.4mm per month at NGP compared to 94.0mm per month at J4M8). That said, the average number of events (25-26 events) and average cumulative rainfall durations (ADD) (650-680 hours) are near- equivalent.

In conjunction with rainfall data, the stormwater flow through each of the SuDS assets has been monitored. Table 4.4 presents a summary of the discharge (m^3/s), velocity (m/s) and flow depth (m) for each asset.

Table 4.4 Flow data summary for the monitoring period

	***J4M8 monitoring period** (January – December 2014)*									***NGP monitoring period** (January – June 2015)*		
	Wetland (J4M8)			**linear wetland (J4M8)**			**Swale (J4M8)**			**Pond (NGP)**		
	Flow (m^3/s)	Velocity (m/s)	Depth (m)	Flow (m^3/s)	Velocity (m/s)	Depth (m)	Flow (m^3/s)	Velocity (m/s)	Depth (m)	Flow (m^3/s)	Velocity (m/s)	Depth (m)
week 2	0.26	0.11	0.87	0.19	0.34	0.45	0.23	0.51	0.33	1.31	0.14	0.46
week 4	0.24	0.11	0.85	0.22	0.35	0.49	0.34	0.60	0.40	1.24	0.14	0.44
week 6	0.21	0.10	0.83	0.25	0.36	0.52	0.28	0.55	0.37	0.98	0.13	0.43
week 8	0.37	0.13	0.94	0.22	0.35	0.48	0.31	0.53	0.41	0.87	0.12	0.42
week 10	0.77	0.17	1.14	0.31	0.39	0.58	0.31	0.51	0.43	0.82	0.12	0.42
week 12	0.56	0.16	1.04	0.24	0.36	0.51	0.24	0.47	0.37	0.86	0.12	0.42
week 14	0.56	0.16	1.04	0.24	0.36	0.51	0.33	0.53	0.43	1.26	0.14	0.44
week 16	0.63	0.17	1.08	0.27	0.37	0.54	0.17	0.40	0.32	1.02	0.13	0.43
week 18	0.48	0.15	1.01	0.19	0.34	0.45	0.19	0.43	0.33	1.12	0.13	0.43
week 20	0.32	0.13	0.91	0.20	0.34	0.47	0.15	0.38	0.30	0.81	0.12	0.42
week 22	0.80	0.17	1.16	0.24	0.36	0.51	0.32	0.54	0.41	0.75	0.12	0.42
week 24	0.53	0.16	1.04	0.18	0.33	0.44	0.29	0.56	0.38	0.68	0.11	0.41
week 26	0.60	0.16	1.06	0.24	0.36	0.51	0.09	0.32	0.23	-	-	-
week 28	0.54	0.16	1.04	0.16	0.32	0.41	0.07	0.29	0.19	-	-	-
week 30	0.28	0.12	0.89	0.18	0.33	0.44	0.06	0.27	0.19	-	-	-
week 32	0.38	0.14	0.95	0.29	0.38	0.56	0.11	0.34	0.25	-	-	-
week 34	0.45	0.15	0.97	0.24	0.36	0.51	0.08	0.29	0.22	-	-	-
week 36	0.26	0.11	0.87	0.29	0.38	0.56	0.09	0.34	0.23	-	-	-
week 38	0.60	0.16	1.07	0.10	0.28	0.32	0.14	0.37	0.28	-	-	-
week 40	0.46	0.15	1.00	0.25	0.36	0.52	0.14	0.40	0.27	-	-	-
week 42	0.24	0.11	0.86	0.25	0.36	0.52	0.23	0.52	0.34	-	-	-
week 44	0.25	0.11	0.86	0.27	0.37	0.54	0.20	0.49	0.31	-	-	-
week 46	0.67	0.17	1.10	0.38	0.41	0.64	0.64	0.79	0.53	-	-	-
week 48	0.58	0.16	1.06	0.09	0.27	0.30	0.51	0.70	0.49	-	-	-
week 50	0.71	0.17	1.11	0.31	0.39	0.58	0.48	0.67	0.48	-	-	-
week 52	0.59	0.16	1.07	0.17	0.32	0.42	0.25	0.54	0.35	-	-	-
Average	0.47	0.14	1.00	0.23	0.35	0.49	0.24	0.48	0.34	0.98	0.13	0.43
Minimum	0.21	0.10	0.83	0.09	0.27	0.30	0.06	0.27	0.19	0.68	0.11	0.41
Maximum	0.80	0.17	1.16	0.38	0.41	0.64	0.64	0.79	0.53	1.31	0.14	0.46

The monitored flows from the field SuDS assets have been summarised in Table 4.4. In terms of average depth of flow, the hierarchy is wetland (1.00m) > linear wetland (0.49m) > pond (0.43m) > swale (0.34m); however the ephemeral nature of the swale led to more highly variable flow depths (up to 0.53m), compared to perennially flowing assets. Further, analysis of flow velocities within the wetland and pond were comparable at low velocities of attenuated range (0.17 and 0.14m/s maximums respectively); this is distinct from the significantly higher and more variable velocities recorded in the swale and liner wetland (up to 0.79 and 0.41 respectively). Yet, despite the low velocity and depth, the total flow discharge within the pond was around twice that of any other monitored SuDS assets (average flow of $0.98m^3/s$), due to high total volume storage and continual flow outlet design.

4.3.2 Suspended and deposited sediment within SuDS assets

The Total Suspended Solid (TSS) and deposition mass for each sample location (Section 3.3) was also monitored using the analytical methodology of Section 3.4.1. Results for each SuDS asset are shown in Figure 4.5, including statistical averages (mean) and standard deviations.

Over the monitoring period the (area weighted) average concentration of suspended solids within the J4M8 SuDS assets (Figure 4.5 a and c) were greatest within the linear wetland (average = 196mg/l), and lowest within the grassed (long) swale (average = 60mg/l). Perennially wet assets (linear wetland, wetland and J4M8 pond) illustrate the greatest range of average TSS concentrations, including data for very low concentrations of TSS (<10mg/l and <1mg/l) possibly reflecting particle settling in these slow-flowing large water bodies; this is less evident for the NGP pond likely due to higher TSS loadings from the construction site.

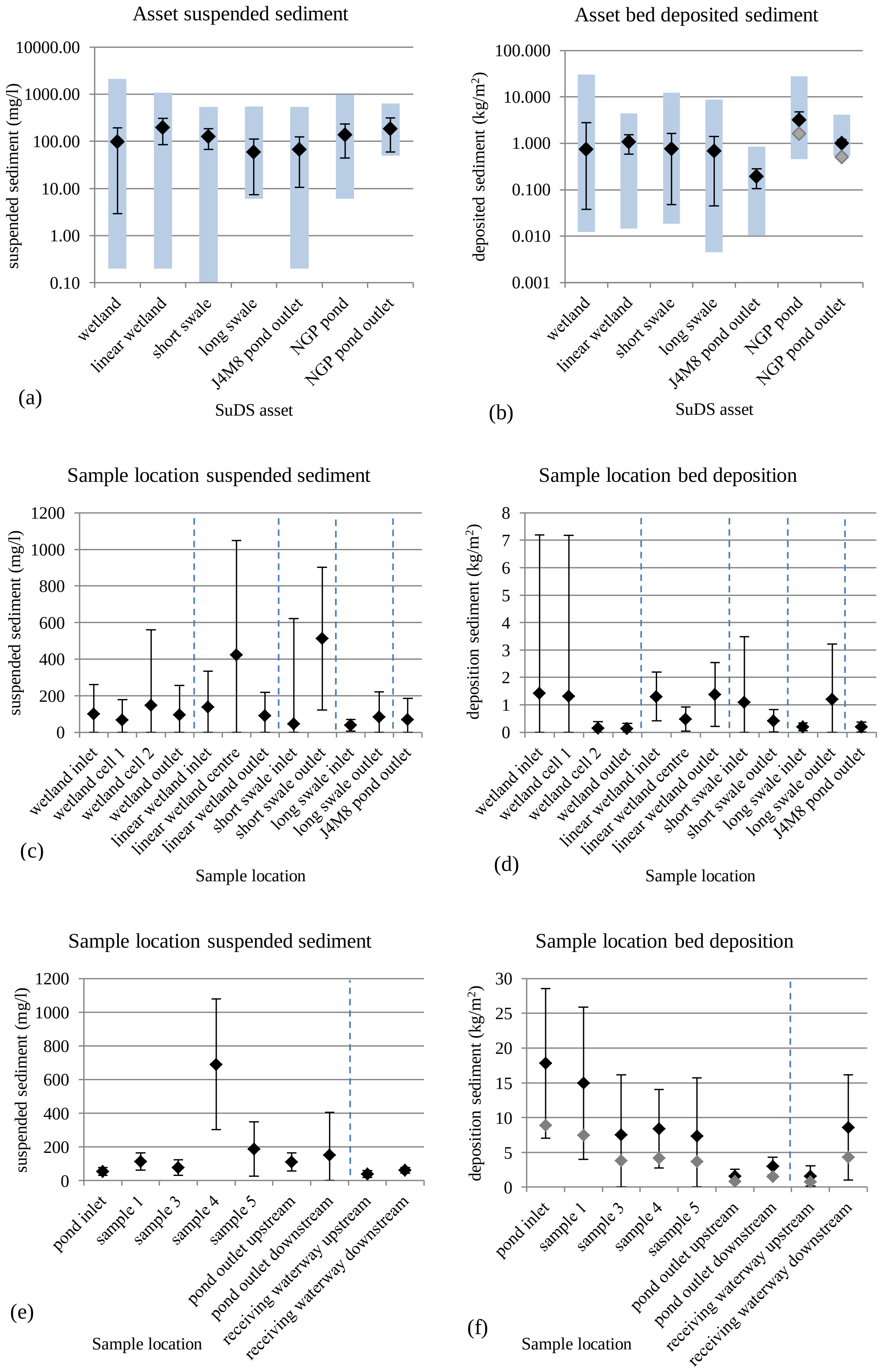


Figure 4.5. Surface TSS concentrations (asset average (a), J4M8 SuDS asset sample location specific (c), NGP pond sample location specific (e)) and bed deposition mass (asset average (b), J4M8 SuDS asset sample location specific (d), NGP pond sample location specific (f)) for the relative sample period. The range is illustrated by the blue bars in the Asset figures (a and b), and standard deviation by the black

lines. Average values and standard deviations are presented to emphasize the asset specific changes and trends in suspended and deposited sediment. The sampling frequency for J4M8 and NGP are fortnightly and monthly respectively. To allow for direct comparison, the approximate fortnightly sediment deposition averages for NGP are presented (grey points in b and f). The location of the sample sites are presented generally in Figures 3.1-3.3 and in greater detail in Figures 5.2, 5.9, 5.15 and 5.21.

It appears from Figure 4.5a that vegetation density within the main flow path may influence TSS concentrations within SuDS assets. The visually observed vegetation density across all monitored SuDS assets follows the order: linear wetland (dense tall wetland vegetation) > short swale (shrub and grass along flow path) > NGP pond (due to reed growth intrusion) > wetland (vegetation around edges but clear flow path)>long swale (mown grass). The asset average TSS concentrations are noted to follow a similar trend; linear wetland (196mg/l) > NGP (139mg/l) > short swale (126mg/l) > long swale (60mg/l). When the standard deviation results are considered, the linear wetland, short swale and NGP pond illustrate very similar results, followed by the long swale, J4M8 pond outlet and wetland (wetland has a greater standard deviation range but notably overlaps that of the long swale and J4M8 pond). The average asset standard deviation and location specific results less succinctly, yet in general, follow the vegetation density order, suggesting some level of influence on TSS results. When the location specific results are considered (Figure 4.5 c and e) it is evident that the linear wetland, short swale and NGP pond have at least one higher detention location that results in elevated average TSS results. Thus the vegetation may be a TSS detention influence in the NGP sample location 4 as this sample location is a densely vegetated area adjacent to the main flow path. The central linear wetland sample location may be relatively elevated due to vegetation influence in conjunction with limited in/outflow influence (potentially resulting in lower results at the inlet and outlet). The short swale elevated outlet TSS results (illustrated to a lesser degree in the long swale) suggest high flow or rainfall inflow influence, resulting in mixing and turbulence at the inlet and therefore resulting in elevated outlet swale TSS results.

When Figure 4.5 (a) and (b) are compared, the average data of TSS and deposits appear positively correlated; i.e. the higher the TSS, the higher the bed deposits. This either shows that suspended sediment supply rate is a dominant process control in providing material for deposits to accumulate or infers that higher TSS may suggest higher total load where bed load contributes to deposits. The location specific TSS results (Figure 4.5 c –f) suggest that when disaggregated, the internal SuDS asset function shows an inverse correlation between TSS and deposition; greater deposition is generally noted to occur in conjunction with lower TSS results. At a site specific level this could be expected from dry weather (non-event) samples; illustration of sediment settling activities within the SuDS assets at sample location level. When considering mass balance analysis of sediment within a SuDS asset, it could reasonably be expected that greater deposition results may occur in conjunction with lower TSS results. However, as the time sequencing of TSS and bed deposition sampling are not event specific but present an overview of sediment deposition and snap-shot suspended sediment occurrence further detailed analysis on the event influence on sediment deposition/suspension is not possible. To address the link between TSS and bed deposition in greater detail continuous or event monitoring of TSS and deposition is necessary. Thus while the time step and detail of field sampling precudes TSS:bed deposition lead/lag analysis a tentative relationship at overall asset and sample location has been attempted.

The asset average deposition data (Figure 4.5 b) across the J4M8 field site SuDS assets show the linear wetland to achieve the highest average deposition (1.1 kg/m^2), shown by area (m^2). When disaggregated to sample location, the inlet section (inlet and cell 1) of the wetland, short swale inlet and long swale outlet all exhibit deposition rates of approximately 1.3 kg/m^2 ± 0.2 kg/m^2. The linear wetland illustrates multiple sample locations (>50%) above 1.3 kg/m^2 illustrating this asset to be consistent in relative elevated sediment detention. However, it is important to note that the upstream half of the wetland has elevated (with large standard deviation) detention results and both swales show an area of elevated detention. The wetland inlet and cell 1 higher deposition results may occur due to the direct supply (surface source) and or flow dynamics of overland/piped stormwater entering a waterbody (resulting in velocity decrease and deposition potential). Similar influences may result in the short swale inlet elevated sediment detention results. The long swale outlet discharges into the J4M8 pond and thus has a wet downstream boundary condition water level. This may reduce flow velocity at the outlet section of this swale and correspondingly result in the elevated sediment detention at this location. Further detailed discussion on deposition influences are presented in conjunction with REO tagged sediment analysis (section 5.3-5.6). The overall J4M8 asset hierarchy (based on average deposition) is linear wetland > wetland ~ short swale > long swale. A further, more detailed analysis of SuDS asset deposition has been undertaken use REO trace sediment (presented in Chapter 5), Through use of the REO tracer the SuDS asset deposition detailed multiple event analysis of monitored SuDS assets have been undertaken to investigate which asset(s) may demonstrate the greatest accumulation.

When the pond outlet results are considered, the NGP pond outlet illustrates much higher TSS and deposition results than the J4M8 pond (again, in response to higher loading from construction source type). Using the data for the NGP pond, it suggests outlet data as 1.0-3.2 kg/m^2 bed deposition occurring within the pond, yet a significant discharge of >100mg/l TSS (but with a large standard deviation range indicating notable variance) discharging from this SuDS asset into the local watercourse. Consideration of location specific deposition within the NGP pond suggests a general declining trend through the pond (both in sample location average and general standard deviation range). This trend, is not reflected in the TSS results, possibly due to the difference in sampling methods (bed deposition sampling presenting total deposition for the preceding month, TSS samples providing a dry weather snap shot of sediment in suspension). The greatest deposition occurs at the inlet, similar to the J4M8 wetland, suggesting effective general detention design of the pond (i.e. detention is occurring throughout the pond not just in one location, e.g. at the inlet or outlet). However, while detention is illustrated within the NGP pond, receiving waterway results do suggest the potential release of sediment from the pond to impact on downstream water quality (TSS and bed deposition). This tentatively suggests that while the pond is function as a sink for urban sediment from the upstream development, it may also be acting as a source of sediment to the receiving waterway. REO tracer monitoring and analysis presented in Chapter 5 (section 5.6) presented further detailed discussion on monitoring location specific deposition and detention of urban sediment within this pond.

4.3.3 Sediment detention capacities of SuDS assets and networks

To provide insight into low flow transportation of fine sediment within the SuDS network, the inflow- outflow TSS concentrations (asset and network) were analysed. Below provide graphical construction to help illustrate monitored SuDS asset functionality (Figure 4.6). This presentation method provides a straightforward visual assessment of low flow SuDS sediment TSS and deposition shift between inlet and out for assets and networks, provided in Figure 4.7 (a – l).

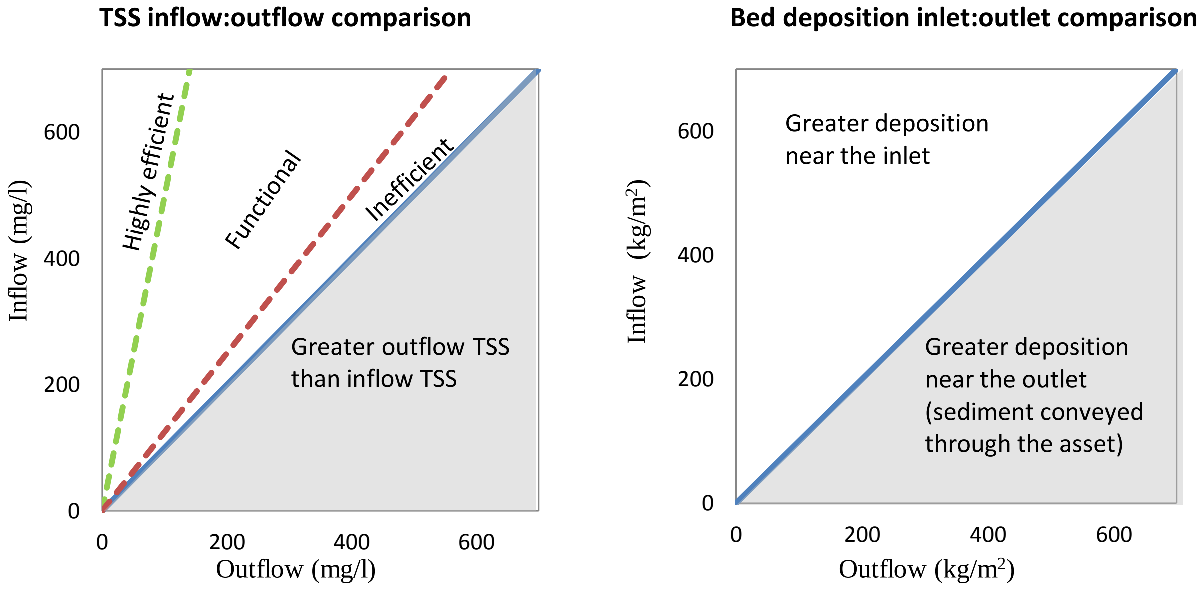
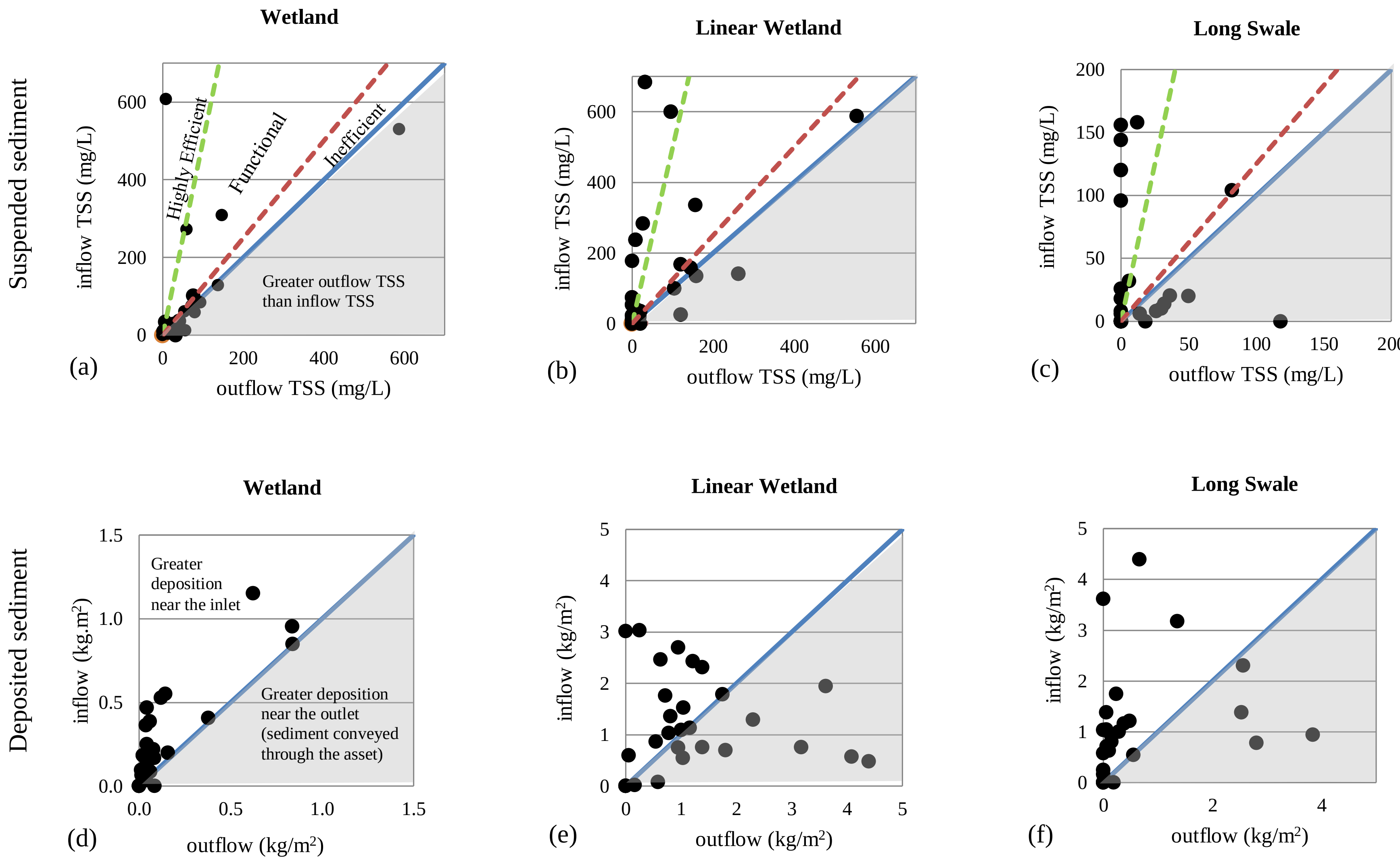


Figure 4.6 Schematic of sediment transport graphical representation as suspended sediment concentration. The inflow and outflow TSS values are from fortnightly/monthly snapshot monitoring (Section 4.3.2). Threshold bounds can be explained as follows: ≤ 0% reflects a net balance of sediment or scour loss of detained sediment via resuspension; 0-20% detention efficiency is classified as "inefficient" water quality improvement; 20-80% TSS removal is arbitrarily considered 'functional' as it lies below ideal design standards but provides some level of water treatment; ≥ 80% is an 'ideal design event efficiency' (Chapter 2, Section 2.4), Bed deposition graphs present only one threshold, the 1:1 line illustrating where greater deposition is found to occur at the inlet or outlet end of the SuDS asset.

Within Figure 4.6 (and the following figures, Figures 4.7, 4.8 and 4.9) three thresholds have been illustrated within the TSS figures and analysis. The 80% threshold is selected as a representation of the ideal SuDS sediment detention efficiency, taken from published literature summarised in Chapter 2, Section 2.6. The 0% threshold is included to illustrate when SuDS efficiency changes from sediment detention to conveyance. The 20% threshold has been selected to demonstrate the lower bounds of when a SuDS asset may be considered functional. Within industry (Local Authority and development engineering), 20% efficiency has been discussed and informally used as a threshold below which a SuDS asset required significant maintenance or remediation[1]. While in fluvial modelling (both sediment and flood modelling) a sensitivity of ±10% is often adopted; in field monitoring analysis variation of ±10% is considered acceptable, adoption of the high 20% threshold in this analysis provides a more conservative functional threshold, informed through professional practice.

[1] Professional experience within Logan City Council and Brisbane City Council, Covey & Associates, WSP, Worley Parsons and Wardell Armstrong LLP.


Suspended sediment
Wetland
Linear Wetland
Long Swale
inflow TSS (mg/L)
outflow TSS (mg/L)
Highly Efficient
Functional
Inefficient
Greater outflow TSS than inflow TSS
0
200
400
600
50
100
150
(a)
(b)
(c)
Deposited sediment
Wetland
Linear Wetland
Long Swale
inflow (kg.m²)
inflow (kg/m²)
outflow (kg/m²)
Greater deposition near the inlet
Greater deposition near the outlet (sediment conveyed through the asset)
0.0
0.5
1.0
1.5
1
2
3
4
5
(d)
(e)
(f)

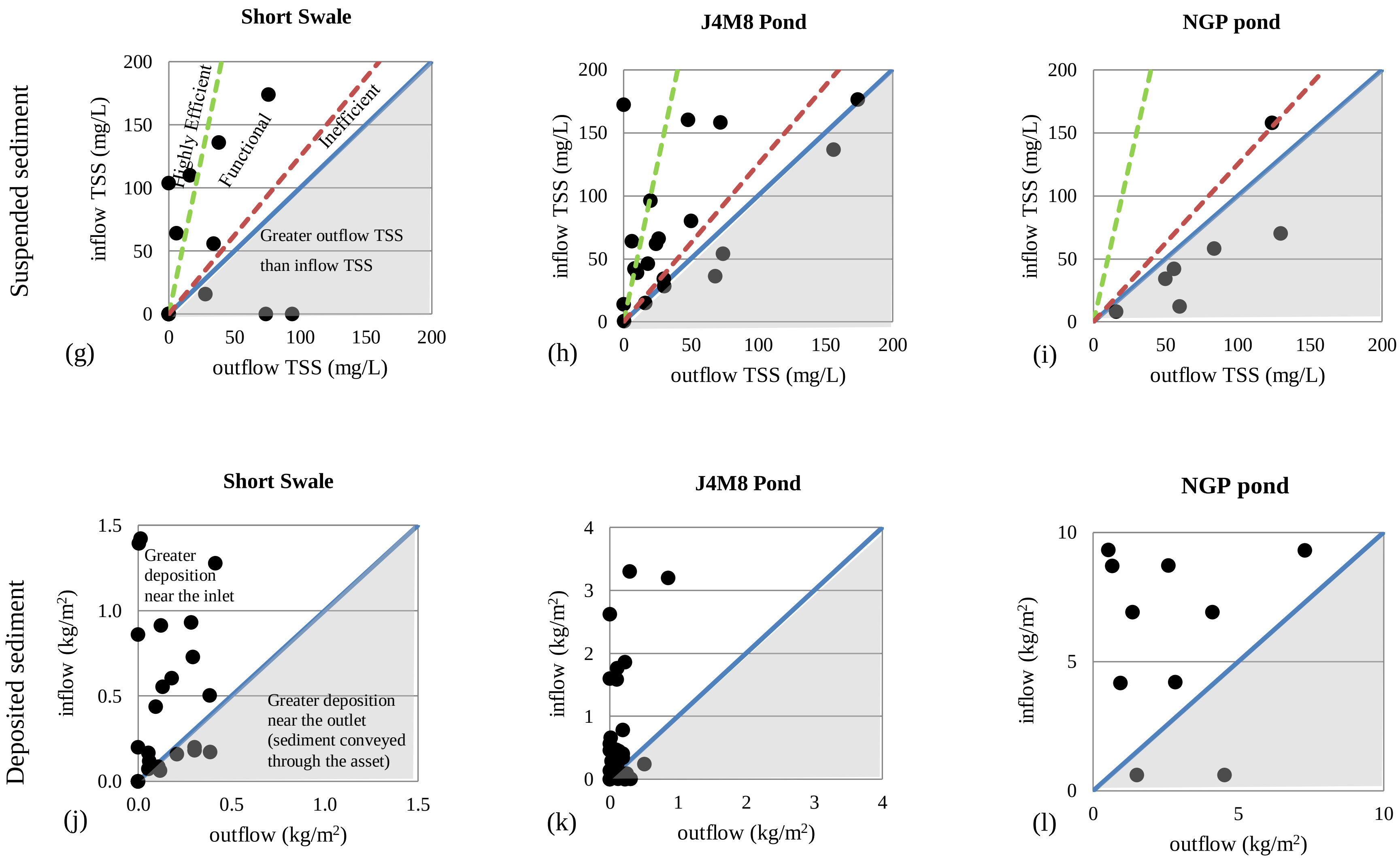


Figure 4.7 SuDS asset suspended sediment and bed deposition sediment detention in dry weather flow conditions. The inlet and outlet suspended sediment and bed deposition monitoring results have been plotted to allow comparison of the influent and effluent sediment. Sample locations are defined in Chapter 3, Section 3.2.

The graphs in Figure 4.7 illustrate the SuDS asset suspended and bed deposition during low flows, commensurate with sampling timing. Two general findings are apparent: firstly, the similarity of TSS and bed deposit trends of Figure 4.5 is weaker in this higher-resolution data set, with the wetland, linear wetland and J4M8 pond most comparable; secondly, all SuDS assets fail to consistently comply with the "highly efficient" design reports or guideline expectations, rather transcending the full range of detention efficiencies and exhibiting (re)entrainment processes via scour. Specific assessment of each asset is provided below:

Wetland: For TSS, the wetland appears inefficient at sediment detention when inflow concentrations are low (<200mg/l); detention appears far more efficient for higher concentrations. Conversely, a higher recurrence (90% data) of bed deposition and detention is observed closer to the inlet, with 50% of the samples recording ≥80% detention near the inlet rather than after conveyance through the SuDS asset to the outlet. The hydraulic attenuation of flow velocity, as it passes into a large volume storage pond (wetland), therefore appears more effective at settling larger particle sizes. Whilst fine sediment may held in suspended load for conveyance through the asset, there is only limited evidence of minor flushing or scour processes.

Linear wetland: The linear wetland appears generally effective in suspended sediment detention with 58% of data show detention ≥20% TSS detention; crucially, 21% of samples achieve high efficiency ≥80% indicating asset capability to perform at design or reported e, transcending the full range of TSS concentrations (Figure 4.7). Whilst asset detention at the inlet is slightly weaker relative to the outlet for bed sediments (47% of samples ≥ 20% at the inlet; 17% of samples were ≥ 80% at the inlet), more significant is the far greater potential conveyance recorded in bed deposit outflow data (up to 4.6kg/m^3). In explanation, the emergent vegetation increases flow path resistance to encourage TSS settling for detention, yet may also contribute to enhanced turbulence for bed scour during higher rates of semi- continuous flow. Thus, linear wetland sediment detention appears related to localised eddy and dead zone influences associated with vegetation stems within the flow path (Huang et al. 2015). In short, the results presented in Figure 4.7 (e) suggest the linear wetland may be highly susceptible to re-entrainment of bed material.

Swale: Overall the swales show a functional TSS sediment detention, with 64% and 90% of data indicating detention ≥ 20% in the long and short swale, respectively. Long swales indicate a di-polar dataset, with TSS detention either 'highly efficient', ≥80%' (57% data, covering the full range of TSS concentrations) or conveyance, ≤0%; the results of the short swale are more variable. As swale design is similar, it appears that increased length (3x) and poorer vegetation maintenance of the long swale leads to visible evidence of sparse vegetation cover which, under extensive conditions of channelized flow, relates to increased bed scour. Hence, greater sediment detention may be achievable by the short swale than the alternative long design.

Pond: Data on Figure 4.7 clearly show both pond SuDS assets to detain sediment closer to the inlet of the pond than the outlet (detention of material rather than conveyance through the SuDS asset) (65-75% of data show greater detention at the inlet relative to the outlet). Both the NGP and J4M8 ponds show the majority of deposited sediment data points to illustrate a 20% preference to the upstream inlet deposition location, suggesting that 20% more sediment is deposited at the inlet end of the pond relative to the outlet. However, this particular type of SuDS asset appears to be of more limited benefit for suspended sediment detention (e.g. 14% TSS data detained in the NGP pond). Further, there appears clear distinction in the efficiency between sites; the J4M8 pond is far more effective at detaining TSS (75%) and exhibits greater 'high efficiency' in bed sediment detention. This may result from: (i) the much larger J4M8 pond size (16,240m^2, versus 2,400m^2 at NGP and depth (>1m, versus >0.5m at NGP), resulting in longer hydraulic residency and higher potential for sediment settling, and/or (ii) the mechanical discharge control (metal valve) maintaining a smaller discharge cross section area to restrict flow release, i.e. a physical barrier to sediment conveyance; this is distinct from NGP's gabion outlet structure where TSS can be more easily discharged.

Networks: Section 3.2 describes the three SuDS networks have been monitored at J4M8 and the stormwater treatment train within NGP. Networks 1, 2 and 4 all include at least one perennially wet SuDS asset (wetland or pond). For the purpose of this Chapter, the network analysis excludes the J4M8 pond as it is the downstream asset common to N1-N3; this permits more detailed analysis of specific differences in network performance due to ephemeral only (N3) versus perennial-ephemeral combinations (N1,N2).

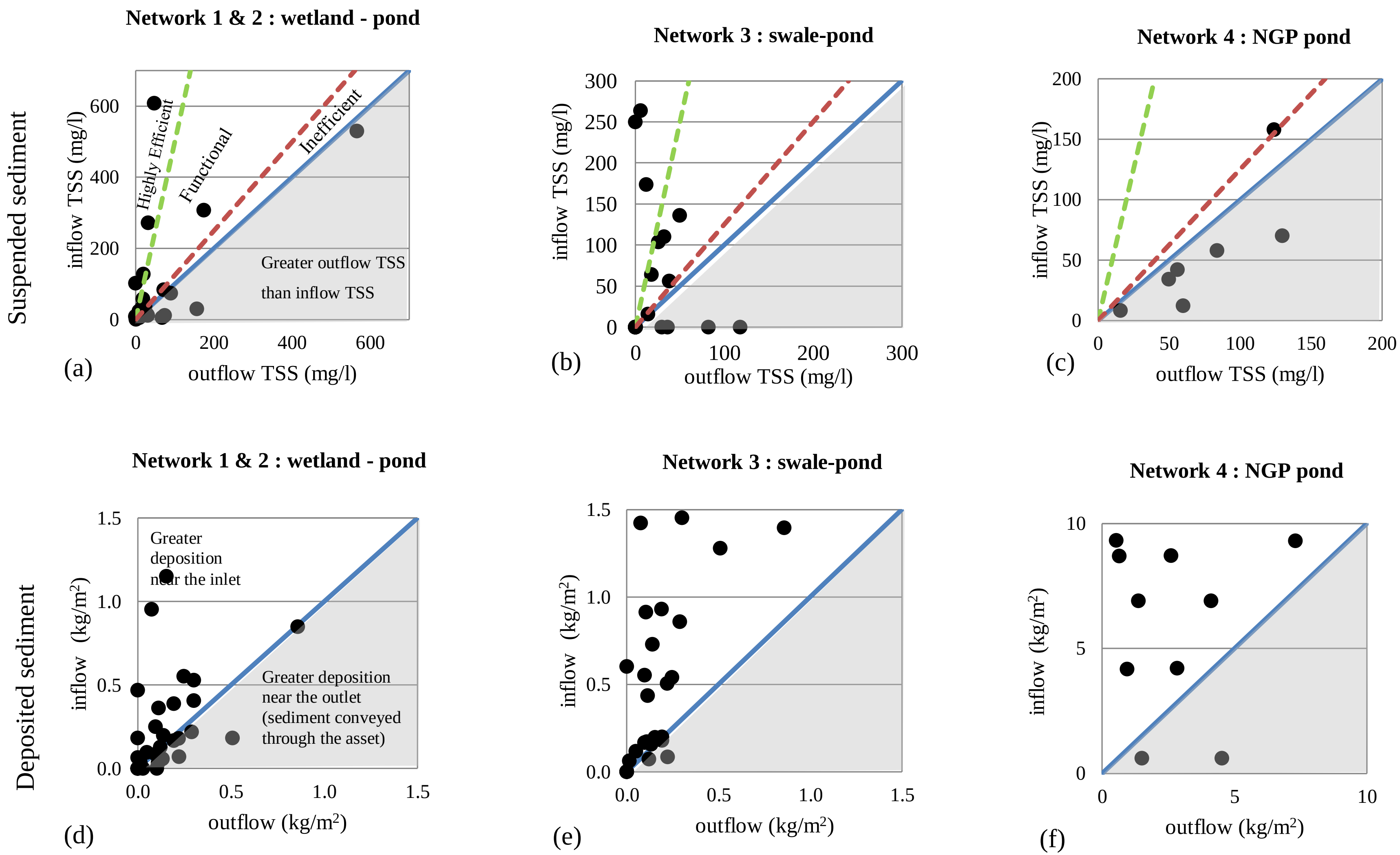


Figure 4.8 SuDS Networks suspended sediment and bed deposition sediment detention in dry weather flow conditions. The inlet and outlet suspended sediment and bed deposition monitoring results have been plotted to allow comparison of the influent and effluent sediment. Sample locations are defined in Chapter 3, Section 3.2.

Figure 4.8 illustrates that N3 (swale – pond) illustrates the greatest dry weather flow TSS detention efficiency, with >90% of results illustrating greater than 20% efficiency (58% of results achieving detention efficiency of greater than 80%). N1 & N2 are second most effective at suspended sediment detention (54% of results greater than 80% efficiency; 69% of results greater than 20% efficiency). Network 4 is found to have the least effective TSS detention results; one sample illustrating a beneficial (>20% detention efficiency) result. These results suggest that networks with ephemeral assets (swales) are considered more effective in TSS detention.

Bed deposition results for the networks do not follow the same trend as suspended sediment results. There is less disparity between N1, 2, 3 and 4 when bed deposition results are considered. Network 4 illustrates the highest overall design inlet detention (40% of results found >80% detention occurred within the inlet extent of the pond rather than the outlet, whilst 80% of all results exhibit detention a level of preferential deposition bear the inlet, >0%). Networks 1, 2 and 3 present comparable results; with 67% and 65% (respectively) of all results achieving greater deposition at the inlet than the outlet. Comparing Networks 1, 2 and 3, the wet networks (N1 & 2) illustrate marginally greater upstream (inlet) sediment detention comparable to network outlet deposition., but at a lower rate relative to N4 pond results; however N3 shows a greater number of data points than N1&2 (N3: 58%, N1&2: 56%) illustrating significant SuDS network prefential detention within the inlet location relative to the outlet. This suggests that, while wet SuDS assets are important in providing detention >0%, a network of multiple assets (N1, 2 and 3) can provide high efficiency (>80% detention within the inlet/upstream extent of the SuDS flow path) or comparable levels to a single larger hydraulic detention asset (N4 pond) (N1 & 2: 22%, N3: 13%, N4: 30%).

N1 & 2 and N3 show moderate TSS efficiencies and inlet bed deposition, while N4 illustrate high bed detention within the upstream SuDS pond flow path but low TSS results. This suggests that multiple asset SuDS networks provide greater TSS detention efficiency than single assets, event when the asset is a pond (N4). The monitored SuDS networks are not illustrated to operate consistently at guideline expected design or reported detention efficiencies (≥ 80%) during dry weather flows. There is potentially a temporal variability of efficiencies that must clearly be better understood scientifically if design guidance is to be revised realistically over the medium-long term design life of SuDS; hence the next sections of this Chapter (and Chapter 6) focus on the process controls.

4.3.4 Factors that may influence total sediment movement

Sediment movement is often mathematically characterised or calculated using factors including flow velocity, stream power and shear stress (Chapter 2, Section 2.6 and Table 2.8). Literature review suggests that the most common factors used to consider total sediment transport capacity are flow velocity, flow depth, shear stress and particle size (Chapter 2, Section 2.5.4). To ascertain why specific SuDS assets function more or less effectively, and to understand the variability in performance within each asset, correlation analysis of the possible influencing factors was undertaken for the monitored dry weather flow suspended sediment and bed deposition. Both rainfall and flow was monitored in conjunction with sediment movement at the NGP and J4M8 field site. Thus, both the flow derived factors and factors describing rainfall occurrence and intensity have been considered in the correlation analysis. Tables 4.5 - 4.7 present the correlation coefficient values for the considered range of factors, from rainfall characteristics to stream power and particle size (parameters defined in Chapter 2 Table 2.8) relative to snapshot TSS and fortnight/monthly cumulative bed deposition within the monitored SuDS assets. Given the natural variability in field data and asset response (Figure 4.7 a-l), high correlation coefficients (>0.8) are unlikely. Instead, this analysis focuses on moderate correlation coefficients (0.5-0.8) as demonstrating a notable relationship; minor coefficients (0.2-0.5) are explored only if there are justifiable scientific assumptions of their significance to the factor demonstrating moderate correlation.

Correlation analysis across flow, particle size *and* rainfall parameters have been undertaken for each monitored SuDS asset overall (Table 4.5), rather than per suspended or deposited sediment monitoring location. This is due each SuDS asset having only a single flow monitoring point, and thus the overall, area weighted (Appendix V) suspended or deposited sediment for each SuDS asset has been assessed in this correlation assessment. While individual sample locations may have provided a more detailed analysis, the generalisation in flow characteristics throughout the SuDS asset was considered unacceptable given the design function of SuDS to slow, detain and decrease stormwater runoff. Thus, a single suspended or deposited sediment value was created for each monitoring period for each SuDS asset. This has been achieved by taking the raw sample data as representative for the cell, calculating the representative total bed deposition and TSS mass (in the water volume at the time of sampling) and then undertaking the correlation using this total SuDS asset TSS or bed deposition value. As a result, each correlation data point within Table 4.5 is based on 26 data points for the wetland, linear wetland and swales and 6 for the NGP pond. All values in Table 4.5 are significant (p-values <0.05) unless specified (indicated by grey text).

Supplementary to this (overview) SuDS asset correlation analysis, an analysis of suspended sediment and bed deposition results at individual sample locations have been correlated with rainfall and modal particle size data (Tables 4.6-4.7). Each correlation data point within Table 4.6 and 4.7 is based on 26 and 6 data points respectively. This has been undertaken as location specific modal particle size data is available and rainfall is directly applicable to all monitoring locations whereas flow information is specific to the sampling location (one sample site per SuDS asset). A limitation of looking at rainfall to sediment sample relationships at this time step is the potential to miss possible local flow patterns influence (i.e. event specific influence). The purpose of this additional rainfall : sediment correlation analysis is to consider whether deposition at individual locations (Figure 4.5) is sensitive to rainfall and particle size characteristics.

Table 4.5 Influencing factors of total sediment detention within the monitored SuDS assets

	Suspended Sediment						Bed Deposition					
	wetland	short swale	long swale	linear wetland	J4M8 pond	NGP pond	wetland	short swale	long swale	linear wetland	J4M8 pond	NGP pond
Flow regime factors												
cumulative flow volume (m^3)	0.3	-0.6	-0.2	-0.2	0.5	0.4	-0.1	0.0	-0.4	-0.5	0.0	0.0
Van Rijn settling velocity	0.1	0.4	-0.1	-0.1	-0.3	0.1	-0.1	0.1	-0.1	0.1	0.0	-0.1
Stokes settling velocity	0.1	0.4	0.0	-0.1	-0.2	0.1	-0.3	0.1	0.0	0.1	0.1	-0.1
water depth (m)	0.4	-0.4	-0.2	-0.2	0.4	0.1	0.2	-0.5	-0.2	-0.3	0.0	0.3
shear stress	0.4	-0.4	0.3	0.1	0.5	0.1	0.0	-0.4	-0.2	-0.6	-0.0	-0.3
flow velocity (m/s)	0.4	-0.4	-0.2	-0.2	0.5	0.2	0.1	-0.5	-0.2	-0.3	-0.0	-0.2
flow (Q, m^3/s)	0.3	-0.4	-0.1	-0.2	0.4	0.4	0.2	-0.2	-0.1	-0.7	0.0	0.1
Stream power	0.3	-0.4	-0.1	-0.2	0.4	0.4	0.6	0.4	-0.2	-0.2	0.0	0.1

Correlation coefficients in *italics* are strong (≥0.8), values highlighted in red are moderate (0.5-0.8) while results highlighted in blue are minor (0.2-0.5). Values identified in grey are not statistically significant (have a p-value > 0.05). This visualisation of correlation values is used in Figures 4.6 and 4.7 for consistency.

Table 4.6 Correlation of rainfall and particle size characteristics with NGP pond sediment at sample locations

	Suspended sediment							Bed deposition						
	pond inlet	sample 1	sample 3	sample 4	sample 5	pond outlet upstream	pond outlet downstream	pond inlet	sample 1	sample 3	sample 4	sample 5	pond outlet upstream	pond outlet downstream
Rainfall factors														
total number of rainfall events since last sample	0.1	-0.5	-0.4	-0.1	-0.5	-0.4	0.4	-0.4	-0.5	-0.4	-0.3	-0.6	-0.6	0.2
ADD prior to sampling (hr)	0.5	-0.1	-0.4	-0.5	-0.1	-0.4	-0.4	0.4	-0.3	-0.4	-0.5	-0.5	-0.5	-0.3
fortnightly monitoring period ADD (hr)	0.3	0.1	-0.3	-0.2	-0.2	0.1	-0.3	0.3	-0.3	-0.5	-0.4	-0.6	-0.5	-0.1
rainfall intensity - event prior to sample	-0.7	0.2	0.7	0.1	-0.1	0.3	-0.3	-0.5	-0.5	-0.3	0.1	-0.4	-0.5	0.5
average event specific rainfall intensity over sample period	0.0	0.4	0.4	-0.2	0.4	-0.1	0.5	-0.2	0.4	-0.5	0.4	0.4	-0.4	-0.2
rain depth - event prior to sampling	-0.6	0.1	0.7	-0.2	-0.3	0.3	0.1	-0.6	-0.2	0.1	0.1	0.1	-0.5	0.4
rain depth - sample period	-0.3	-0.3	0.3	-0.5	-0.1	0.3	0.7	-0.2	-0.3	-0.7	-0.3	-0.5	0.3	0.4
Particle size factors														
SS modal particle size	-0.3	-0.4	-0.3	-0.4	-0.1	-0.1	-0.3	0.1	-0.4	-0.2	-0.3	-0.4	-0.3	-0.4
Bed deposition modal particle size	0.3	0.4	0.2	-0.1	-0.4	-0.2	-0.4	0.1	0.2	0.3	0.3	0.2	0.2	0.4

Table 4.7 Correlation of rainfall and particle size characteristics with J4M8 SuDS sediment at sample locations

	Suspended sediment												Bed deposition											
	wetland inlet	wetland cell 1	wetland cell 2	wetland outlet	linear wetland inlet	linear wetland centre	linear wetland outlet	short swale inlet	short swale outlet	lone swale inlet	long swale outlet	J4M8 pond outlet	wetland inlet	wetland cell 1	wetland cell 2	wetland outlet	linear wetland inlet	linear wetland centre	linear wetland outlet	short swale inlet	short swale outlet	lone swale inlet	long swale outlet	J4M8 pond outlet
Rainfall factors																								
total number of rainfall events since last sample	-0.1	-0.3	-0.2	-0.1	0.0	-0.2	-0.4	-0.5	-0.3	-0.1	-0.4	0.1	-0.3	-0.1	-0.1	-0.1	-0.1	-0.3	-0.1	-0.1	-0.2	-0.1	-0.1	-0.1
ADD prior to sampling (hr)	-0.3	-0.1	-0.2	0.0	-0.1	-0.1	-0.2	-0.3	-0.1	-0.3	-0.2	-0.3	-0.1	-0.2	-0.1	-0.1	0.0	-0.3	0.0	-0.1	-0.3	-0.2	-0.3	-0.2
fortnightly monitoring period ADD (hr)	-0.4	-0.4	-0.4	-0.2	-0.1	-0.1	-0.3	-0.4	-0.2	-0.4	-0.3	-0.1	-0.4	-0.6	-0.4	-0.3	-0.2	-0.4	-0.3	-0.6	-0.2	0.1	0.2	-0.7
rainfall intensity - event prior to sample	-0.1	0.0	-0.1	-0.2	-0.1	-0.2	-0.2	-0.3	-0.4	-0.2	-0.1	0.2	-0.2	-0.2	0.0	0.0	0.1	-0.2	0.1	-0.2	-0.1	-0.1	-0.1	0.1
average event specific rainfall intensity over sample period	0.1	0.2	0.0	0.0	-0.1	0.0	0.2	-0.3	0.0	0.2	0.1	0.1	0.0	0.0	-0.4	-0.5	-0.2	-0.3	0.2	-0.2	-0.5	-0.4	-0.1	0.1
rain depth - event prior to sampling	0.0	-0.2	0.0	-0.1	-0.1	0.0	-0.2	-0.1	-0.4	-0.2	-0.1	0.0	0.1	0.2	0.4	0.5	0.4	0.1	-0.3	0.1	0.4	0.2	0.3	0.0
rain depth - sample period	0.0	-0.3	-0.2	-0.1	-0.1	-0.2	0.4	0.1	-0.2	0.5	0.4	0.2	-0.2	0.1	0.3	0.3	0.2	-0.2	-0.3	0.3	0.2	0.2	-0.1	-0.1
Particle size factors																								
SS modal particle size	-0.1	-0.2	-0.1	-0.2	-0.2	-0.2	-0.1	-0.3	0.1	-0.4	-0.1	-0.1	-0.1	0.1	-0.2	0.1	0.1	0.1	0.3	0.2	0.3	0.1	0.4	0.1
Bed deposition modal particle size	-0.2	0.4	-0.1	0.2	0.0	0.3	0.4	0.4	-0.1	0.1	0.1	0.1	0.1	0.1	0.3	0.1	0.3	0.3	0.1	0.2	0.3	0.4	0.4	0.2

There are few major or moderate factors (major or moderate correlation results) that influence the overall SuDS asset identified from the field dataset (Table 4.5). In general, suspended sediment detention appears to be asset specific with only the short swale and pond exhibiting notable correlations. Whilst both assets correlate to cumulative flow volume, the pond demonstrates stronger correlations to flow regime factors (velocity, shear stress), whilst the short swale is responsive to the number of rainfall events. This reflects distinction between perennial (flow) versus ephemeral (rainfall) driven assets.

Suspended sediment moderate flow regime correlations in Table 4.5: suspended sediment detention and: flow velocity (J4M8 pond), shear stress (J4M8 pond), cumulative flow volume (short swale, J4M8 pond). There is a moderate positive correlation between pond suspended sediment detention and flow volume. This may be a natural pond (and to a minor extent wetland) function, allowing greater treatment to occur within the central flow path (open water section) when total flow volumes are greatest due to a lateral flushing (forcing sediment laden flow to enter the vegetated wetland boundary areas and become detained). Moderate negative correlation between cumulative flow and the short swale suggests that the ephemeral asset conveys sediment during higher cumulative flows, supporting resuspension and transport through and/or over the more uniform vegetation and through the more uniform lateral asset design.

The J4M8 pond illustrates a positive correlation with both flow velocity and shear stress (a derivative of velocity). A lower flow velocity causes less turbulence, thus providing a greater sediment setting potential. As a result, both flow velocity and shear stress illustrate positive correlations within the J4M8 pond (and wetland), indicating the perennial asset suspended sediment resuspension and conveyance through elevated flow velocities.

Suspended sediment moderate rainfall parameter correlations in Table 4.5: The short swale suspended sediment illustrates a moderate negative correlation with fortnightly rainfall event occurrence. The ephemeral swales negative correlation suggests that more numerous rainfall events result in less sediment in suspension, thus suggesting that the number (or occurrence) of rainfall and flow events is important in the transport of sediment pollution through the ephemeral assets.

When considered at sample location (Table 4.7) both short swale inlet and outlet show moderate negative correlation results to fortnight rainfall event occurrence. This suggests greater TSS detention is recorded when fewer rainfall events occur. There may be an element of hysteresis influencing deposition, that the rainfall events occurring prior to sample collection have a longer duration of influence than the event hyetograph or hydrograph. It may also be reflective of the TSS particle site, as silt and clay material requires an extended duration without turbulence to effectively settle out of the water column. The strength of the correlation is lower at the outlet, suggesting possible flow mitigation influence relative to rainfall event occurrence through the short swale. This may also possibly reflect the generally higher TSS concentrations at the outlet (Figure 4.5) and potentially the ease of fine sediment conveyance through/over the short vegetation and settling velocity (duration) of fine material entering the swales. There may also be a link to supply, as more numerous rainfall events occur, the quantity of sediment washoff from urban surfaces may decline resulting in less sediment conveyance per rainfall event. Thus if a greater quantity of rainfall events have occurred prior to monitoring, beyond a threshold where urban source balances washoff rates, the rainfall may result in 'cleaner' inflow and thus potentially lower overall detention. NGP pond backflow area, central open water section (sample 3) and main flow path (sample 5) also show negative correlations to cumulative rainfall occurrence. This is similar to the short swale, and correlations may occur due to the conveyance of sediment due to multiple rainfall event into and through the pond.

Rainfall depth prior to sampling illustrates the magnitude, without dynamic information, on the preceding rainfall event. A greater rainfall depth may illustrate a longer or more intense rainfall event prior to sampling and could potentially result in greater turbulence and suspended sediment. This may explain the moderate positive correlation found for the centre of the pond (sample 3), while the inlet follows rainfall intensity trends (negative moderate correlation potentially due to flushing, conveyance or to dilution). When the preceding fortnight total rainfall is considered, the moderate correlation shifts to the vegetated area along the flow path in the pond (sample 4) and the outlet (downstream). This suggests the open water area may be more sensitive to more recent rainfall events while the vegetated area and outlet may show a greater response to cumulative events.

Rainfall depth for the preceding sample period (i.e. fortnight at J4M8) illustrates a moderate correlation with the long swale inlet (Table 4.6). The correlation strength is lower at the outlet. This suggests elevated suspended sediment concentrations at the long swale inlet may be influenced by the occurrence greater fortnightly rainfall depth, an indicator of total rainfall. Rainfall depth may function to illustrate general rainfall influence on the long swale, without the nuances of intensity, duration or frequency, and thus in some way allow the overall generalised influence (in whichever form, occurrence, intensity or duration) to be considered on suspended sediment concentrations. This moderate correlation may occur due to greater rainfall resulting in greater flow, thus allowing greater volume of stormwater and urban sediment to reach the long swale (2^{nd} or 3^{rd} in J4M8 Networks 1-3).

The NGP illustrates several moderation correlations between suspended sediment : ADD and rainfall intensity. The ADD period prior to sampling has a moderate correlation to the pond inlet and vegetated deposition area (sample 4). A longer ADD duration prior to sampling could be expected to allow greater sediment settling and therefore lower suspended sediment results. This may explain the negative moderate correlation within the pond (sample 4). The positive correlation at the inlet may be due to the continue inflow occurring at this site, even in dry weather (continuous low inflow due to upstream land use). Thus, the positive moderate correlation at the inlet may illustrate the ongoing mixing and suspension of fine sediment at this location due to continual inflow, and therefore high suspended sediment concentrations during longer ADD durations.

Rainfall intensity illustrates several moderate correlation values relative to the NGP pond. A negative rainfall intensity: TSS correlation suggests that the increased rainfall intensity may be relative to surface washoff , thus with intense rainfall there is a first flush of sediment influx and potential for latter dilution or conveyance (potential explanation for inlet correlation at sample point 3). A positive correlation (greater rainfall intensity occurring in conjunction with elevated TSS results) may be due to mixing or rainfall induced turbulence within the pond, potentially illustrated in the open water section of the pond (sample 3). Furthermore, the positive moderate correlation seen at the pond outlet (downstream) suggests river influence on this sampling location; greater catchment rainfall intensity resulting in greater turbulence and TSS in the river water column.

Further, bed sediment detention data of Table 4.5 clearly show multiple flow regime factors influencing linear wetland results (cumulative flow volume, shear stress, and discharge) which may have correspondence to rainfall depth drivers. The other moderate correlations in bed data indicate potential relevance of rainfall depth and intensity in the short swale (ephemeral) and wetland (i.e. the upstream most asset, hence rainfall responsive), and correspondingly a strong correlation (≥0.8) between cumulative ADD (also noted in the pond).

Bed deposition moderate flow regime correlations: stream power (wetland), flow discharge (linear wetland), flow velocity (short swale), shear stress (linear wetland), flow depth (short swale), cumulative flow (linear wetland).

Negative moderate correlation between cumulative flow volume and linear wetland sediment detention efficiency indicates the influence of cumulative flows on this assets sediment detention. This follows the suspended sediment correlation for the short swale, suggesting that larger cumulative flows result in greater resuspension and transport of fine sediment pollution through the ephemeral asset. Similar to the linear wetland, the short swale illustrates a moderate negative correlation with flow depth, indicating greater flow depths (related to higher flow volumes and discharge) to convey greater sediment through and over the grass swale.

The linear wetland and short swale show moderate correlation to discharge/shear stress and flow velocity (respectively). Both ephemeral assets illustrate higher velocities and discharge to result in conveyance rather than deposition and resuspension of deposited sediment. The stream power, a descriptor of flow force, positive correlation in the wetland suggests that, similar to the pond and wetland influence by larger cumulative flows, greater stream power may have a lateral flushing effect.

Bed deposition moderate rainfall parameter correlations: The wetland and short swale (upstream SuDS assets) illustrates a moderate to strong correlation with rainfall intensity and rainfall depth. Greater rainfall (depth) is linked to greater sediment detention in the wetland and less detention in the short swale. This illustrates the different sediment detention processes in perennial and ephemeral SuDS assets. Rainfall intensity (average intensity over the sample period) has a negative correlation with the short swale and wetland sediment detention (higher intensity rainfall results in less effective detention). Within the wetland, greater flow depth supports greater detention efficiency, potentially due to slower flow velocities resultant from the larger surface area of the perennial asset. The short swale negative correlation suggests that greater rainfall results in greater sediment pollution conveyance rather than detention and deposition.

Cumulative ADD (wetland cell 1, short swale inlet, NGP pond central flow path and outlet; upstream SuDS assets), rainfall intensity depth during event prior to sampling (wetland and short swale outlet, NGP pond inlet and outlet) and rainfall depth (wetland outlet, NGP central flow path and outlet) all show moderate correlation to sediment deposition . The upstream SuDS assets, assets receiving untreated urban stormwater, show a strong negative correlation to cumulative antecedent dry period and recent rainfall intensity. This suggests that larger cumulative ADD concur with lower sediment detention efficiency, and the cumulative duration, as opposed to the ADD prior to sampling, has a strong influence on bed deposition. It also suggests that the wetland bed deposition efficiency may be influenced by individual rainfall events and the resultant influx of sediment/resuspension occurring as a result (as a more descriptive characteristics of events compared to flow velocity or depth change in this ephemeral upstream SuDS asset).

It is noted that the J4M8 pond also shows a moderate positive correlation to rain depth. This may occur due to the placement of sampling location at outlet, downstream from the large pond, and as a result sediment supply out of this pond is greater after more sizable rainfall, allowing greater deposition to potentially occur. A similar anomaly is noted with the NGP outlet downstream results, where the sample is taken downstream of the pond outlet and appears to be influenced by the receiving waterway (both as a cause of turbulence and as a deposition area of upstream river sediment as well as pond conveyed material).

Particle size correlations: Finer particle size sediment can be expected to remain in suspension for a greater time period than coarse sediment. Cohesive sediment, forming floc particulates, may settle faster than individual fine clay or silt particles (up to a threshold, >10kg/m^3 (Mehta 1986) after which settling velocity declines. The occurrence of cohesive sediment within urban stormwater results in additional complexity in the particle size: deposition relationship, resulting in a greater potential for clay and silt material to become deposited during elevated flow conditions.

The modal particle size correlation results for the individual sampling locations do not show any moderate strength correlations. The minor correlations do correspond to the expected relationships, with suspended sediment : modal particle size illustration a generally negative minor correlation and bed deposition : modal particle size illustrating positive minor correlations. The lack of strong correlations suggests that particle size alone, and therefore potentially settling velocity, is not a solo key driver in sediment deposition within SuDS assets but is of influence in conjunction with rainfall and flow parameters (as well as vegetation density and asset design). As a result of this complexity further particle size correlation analysis has been undertaken using the REO tracer sediment dataset and is presented in section 6.1.3.

Several factors are found to be moderately or strongly to specific SuDS assets, but no one factor illustrates a strong influential across all SuDS assets, for all suspended sediment detention or all bed deposition fluctuations. The most significant result from the correlation analysis is illustration that total sediment detention, deposition and transport is a highly complex process, with multiple influential factors influencing SuDS asset sediment detention or conveyance efficiency at any one time. The results also suggest that each asset functions in an individual way, and the total fine sediment detention process cannot be easily generalised for blue-green vegetated SuDS.

Significant detailed analysis of rainfall, flow and particle size characteristics on sediment conveyance and detention within each of the monitored SuDS assets is undertaken and presented in Chapter 6. This more detailed correlation analysis and assessment of SuDS sediment detention is possible due to the REO trace monitoring and sediment dataset created and presented within Chapter 5.

4.3.5 SuDS network influence on detained sediment particle size

Downstream trends in particle size distribution (Figure 4.9) through N1 and N2 (wetland-pond networks) SuDS network shows a decreasing trend in the size of material deposited within the sediment traps and found within the surface water. This supports the use of these specific multiple-asset treatment train designs in progressive removal of finer materials from suspension, such that TSS material <63µm (Figure 4.9) can be detained for treatment; this is crucial in remediation of heavy metal stormwater pollutants commonly associated with <250µm fractions (Jones et al. 2008, Adiyiah et al. 2014). Within these networks (N1, N2), the coarser bed deposits are, therefore, found in the upper assets of the treatment train or network (especially the linear wetland), suggesting local residence and detention efficiency.

Conversely, Network 3 offers a different treatment train asset sequence, constrained to upstream swales and limited to 3 SuDS assets. Neither TSS nor bed deposit data illustrate any significant downstream trend in PSD, implying no selective removal of specific fractions in the swale design. As the bed PSDs are typically finer in N3 (than

N1 or N2) it may be that the vegetated filter design (Section 3.3.2) is successful in the removal of coarser fractions prior to entry of wash-off into the swale (i.e. within the vegetated filter strip); hence these assets only require performance for the finer fractions as reflected in them tending to the same TSS PSD <63µm as N1 & N2.

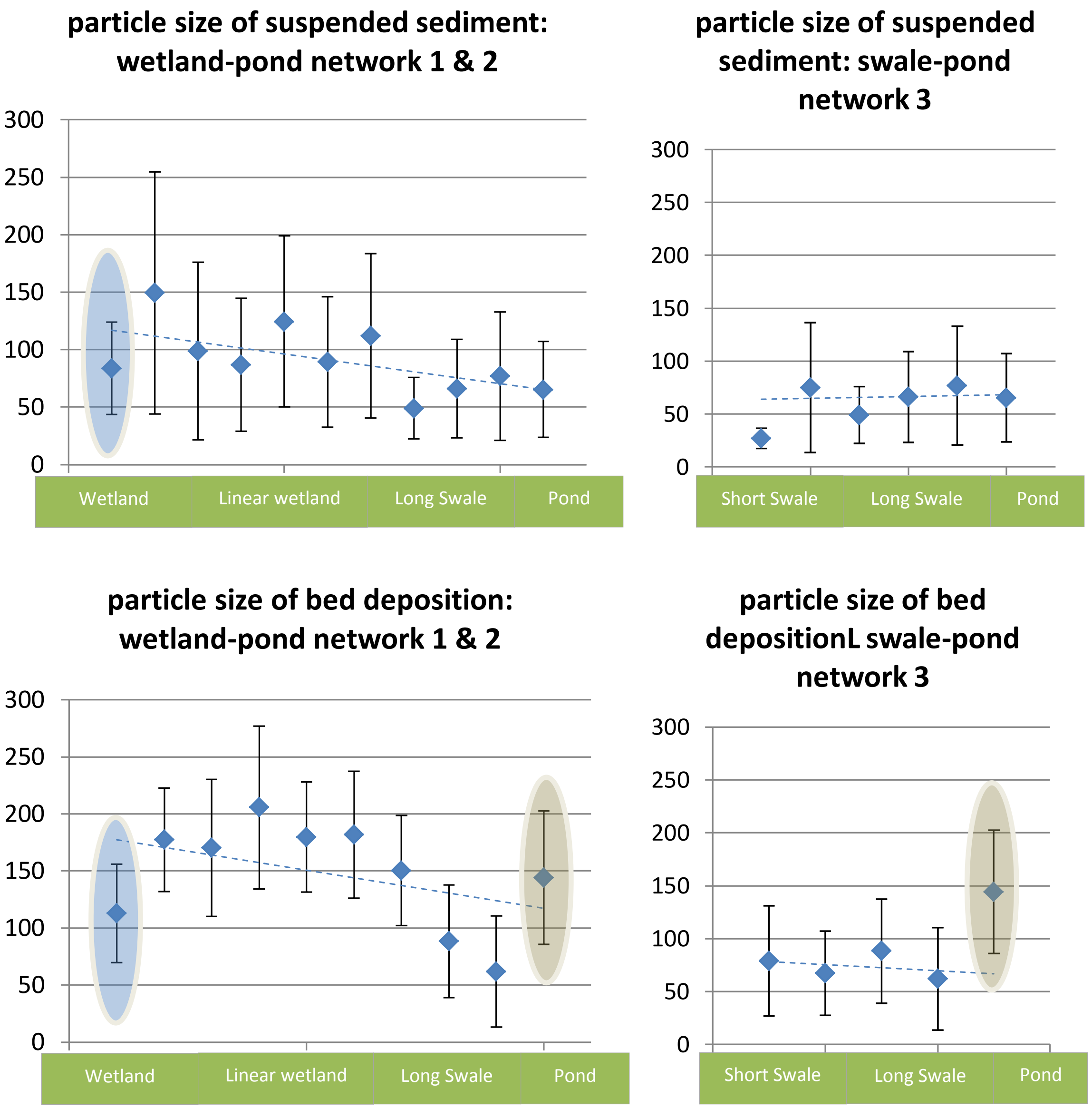


Figure 4.9 Modal particle size of samples the monitored SuDS assets within Networks 1-3. The range of data from all samples collected are provided (black range lines), with the average modal particle sizes highlighted blue (dots). Highlighted (circled) blue outliers with finer than expected sediment are found at the upstream inflow into N1 and N2; this reflects wetland design causing hydraulic attenuation and energy loss as flow enters the high-volume waterbody. The highlighted (circled) grey outlier at the pond inlet of N1 and N2 reflects inlet design and potential influence of turbulent scour at the sample point.

4.4 Chapter Conclusions

The research presented in this Chapter provides new field monitored data (TSS, bed deposits, rate, PSD) for sediment delivery (wash-off) and conveyance within SuDS assets and network. These data were used to calculate and present a range of sediment detention efficiency results for dry weather flows. Correlation analysis has helped define flow and rainfall factors worth further analysis in terms of sediment detention efficiency in Chapter 6. Modal particle size analysis provides insight into SuDS asset and network detention of particle size fractions, important in the further analysis of urban pollution treatment, analysed in Chapter 7. In summary:

- Discrete land-use typologies monitored in this research conform generally to the sediment load and characteristics identified in the wider literature, supporting the field site selection as representative of urban development land use;
- Sediment wash-off rates are fastest for road surfaces and via a piped flow path (Figure 4.3);
- The average bed deposition in the J4M8 networks was greatest within the linear wetland (1.10mg/m^2/fortnight). The suspended sediment concentrations are noted to follow the bed deposition trend, with the highest average suspended sediment results illustrated in the linear wetland (196mg/l). This suggests a link between deposition and suspended sediment, potential related to sediment availability;
- The NGP pond provides the highest average bed deposition (~1.6kg/m^2/fortnight) overall. The NGP pond deposition range is elevated compared to that found in J4M8 SuDS assets but with comparable suspended sediment results. Thus the NGP pond suspended sediment detention is elevated but comparable to J4M8 SuDS, but illustrates notably lower bed deposition detention efficiency than the J4M8 assets;
- SuDS assets and the SuDS networks do not provide a consistent level of sediment detention over the medium-longer term monitoring horizons considered herein. The majority of sediment detention provided by these SuDS assets is less than the expected ~80% design-guidance efficiency threshold (Section 2.6). This infers asset and network underperformance over the medium-long term, when multiple flow events are considered. It cannot be assumed that the quantity or trends in bed and TSS sediment detention are the same; each exhibit distinct responses to process controls and network design;

- Correlation analysis illustrates a distinction between perennially wet assets (pond, wetland) and ephemeral assets. Perennial assets generally indicate inverse correlation response to ephemeral assets, illustrating wet, hydraulic detention SuDS assets to provide sediment detention through differing process and flow/rainfall response than ephemeral assets. However, this conclusion is drawn from moderate correlations only due to the complexity of the natural environment (Table 4.5); and
- The design of the SuDS network can influence the size of sediment detained within the system. The wetland-pond SuDS network (N1 and N2) shows a PSD trend that supports moderate sediment detention within the upstream SuDS and fine sediment detention in the downstream SuDS assets. However the swale-pond SuDS network (N3) appears to receive urban sediment with a smaller modal particle size and shows less change is particle size detention (modal particle size illustrates limited change down the SuDS network).

Thus the research findings present herein illustrates that sediment detention by SuDS assets and networks do not conform consistently to the design standards set out in current guidelines (both nationally and internationally) or when compared to each other. Furthermore, when considering fine sediment, the detention processes within SuDS assets are more complex than settling velocity, hydraulic residency time and shear stress and the influential factors on bed deposition are not identical to those for suspended sediment detention. The complexity in SuDS sediment detention, both the efficiency in asset deposition and detention over multiple rainfall-runoff events, is difficult to analyse as the element of resuspension and continual movement is invisible within this mass deposition dataset. Thus, using the total deposition and detention efficiency presented within this Chapter, the movement of (REO) tagged discrete sediment releases through the SuDS assets is examined in Chapter 5 and 6, extending this mass deposition and efficiency dataset to identify the ongoing movement and driving processes behind SuDS asset sediment detention efficiency.

5 Chapter 5: Sediment transport through SuDS over multiple events

5.1 Introduction

Field studies and numerical modeling have been undertaken in previous studies to calculate and estimate the total suspended sediment and mass of sediment detained within SuDS assets (Chapter 2.6). As this previous analysis has generally either been undertaken as event specific analysis or via total sediment load trend analysis (Table 2.8), existing SuDS design has been based on conceptual processes described via a single design event flow. It is, therefore, unsurprising that the longer term monitoring results presented in Chapter 4 have identified deficiency within these general design and operation assumptions. Specifically, Chapter 4 data illustrates that sediment detention within SuDS assets are neither uniform nor constant over time/multiple events. Consequently, the monitored SuDS networks fail to consistently achieve best practice design guideline expectations or guidance reported values of 80-90% sediment removal rate (Chapter 2.6) when multiple event flows (and therefore extended periods) up to and including design flows (2 year RP rainfall-runoff events) (SWITCH 2011, CIRIA 2015, PSMM 2014). This leads to reasonable concern being raised over the design, operation and performance of current SuDS, which can only be resolved by monitoring SuDS as a white-box (physical processes) system. Thus, the intention of Chapter 5 is to quantify sediment movement within SuDS assets and networks over an extended monitoring period and resultant of multiple rainfall-runoff events.

Crucial to the research presented in this Chapter is the fundamental requirement to trace the movement of unique, defined ‘parcels’ of sediment through a SuDS asset and full network. Chapter 3 describes the development of a novel urban sediment REO tracing method, appropriate for monitoring sediment conveyance, over the course of up to a year, through the four SuDS networks of the present thesis. This Chapter addresses the following key research objectives: i) identification of fine sediment movement within SuDS assets; ii) quantification of the fine sediment conveyance though individual SuDS assets over multiple low flow events; iii) mass-balance analysis to quantify the fine sediment detention over a 6 or12 month multiple rainfall-runoff event period to illustrate resuspension occurrence and overall fine sediment detention.

5.2 Tagged sediment transport through individual SuDS asset

Acknowledging that fine sediment entering a SuDS asset continued to move over multiple rainfall-runoff events (Chapter 4) presents the question: How much sediment moves? To quantify the mass of sediment conveyed through individual SuDS assets, the REO tagged sediment dataset was disaggregated to asset specific parcels. A known quantity of REO tagged material was released into each system, monitored at the inlet through both suspended sediment sampling and sediment trapping on the asset bed (bed deposition) and throughout the SuDS asset. Monitoring of REO tagged material was undertaken to illustrate the redistribution of tagged sediment within the asset and discharge from the asset. Individual sample points within the asset are therefore assumed representative of the wider sample area and data was processed using an area weighting specific to the sample location (or volume weighting for suspended sediment results). This permitted a basic mass balance analysis for each SuDS asset at each monitoring time step, the quantity of tagged sediment detained over multiple events within each SuDS asset and network has been calculated, and is presented in the following sections.

Each SuDS asset was sampled in multiple locations (defined in Chapter 3, Section 3.3). Each sample provided suspended or bed deposited sediment mass representative of the specific sampled area or zone of the SuDS asset, quantified by the REO tag concentration in sample results. The total sediment within each SuDS asset was therefore calculated as the sum of the area-weighted suspended sediment and bed deposition mass.

For each asset, a total REO tagged sediment detention curve has been created by comparing the quantity of REO tagged sediment entering the SuDS asset (either from release off the upstream urban surface or discharge from the upstream SuDS asset) to the quantity of REO tagged sediment detained in the SuDS asset. The detention curve has been creased as a composite of the bed deposition (an area weighted calculation of deposition based on the bed deposition (core) samples) and REO tagged sediment in suspension (suspended sediment samples extrapolated by volume). Figure 5.1 illustrates how these total REO tagged sediment curves were compiled.

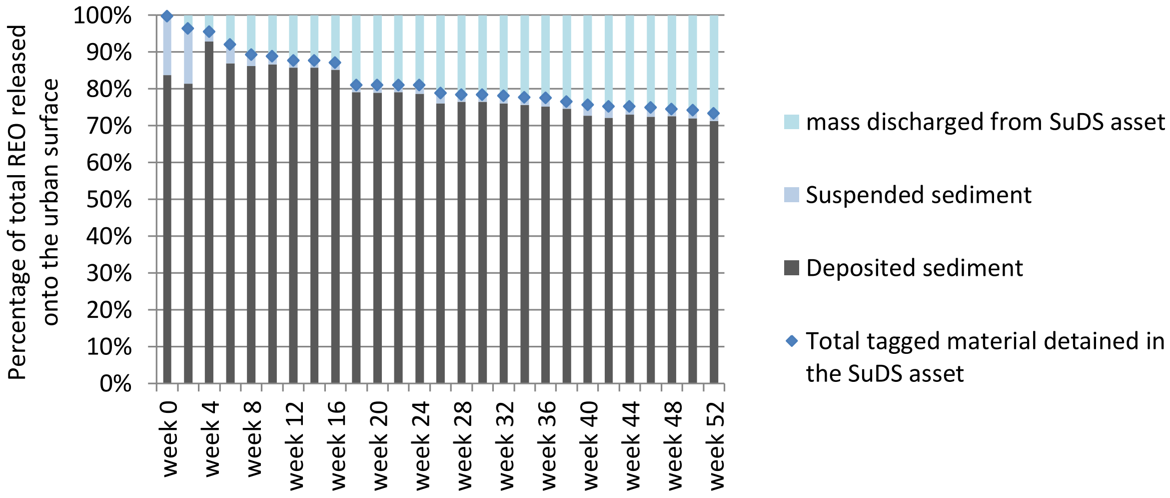


Figure 5.1 Schematic of total REO tagged sediment curve compilation. All REO tagged sediment results presented in the following sections (Section 5.3 – 5.6) follow this analysis methodology.

5.3 Wetland

Tagged sediment was supplied to Network 1 by overland flow. Surface runoff was conveyed from the source surface into the wetland as sheet flow (Figure 5.2). Alternatively, tagged sediment was supplied to Network 2 by pipe flow, collecting and conveying roof stormwater runoff sub-surface prior to releasing it into the wetland. This stormwater pipe discharged into the wetland sub-surface, thus requiring vertical mixing for tagged sediment particles to occur at surface level. The significance of this distinction in design feature, and the associated process controls, is the focus of this section.

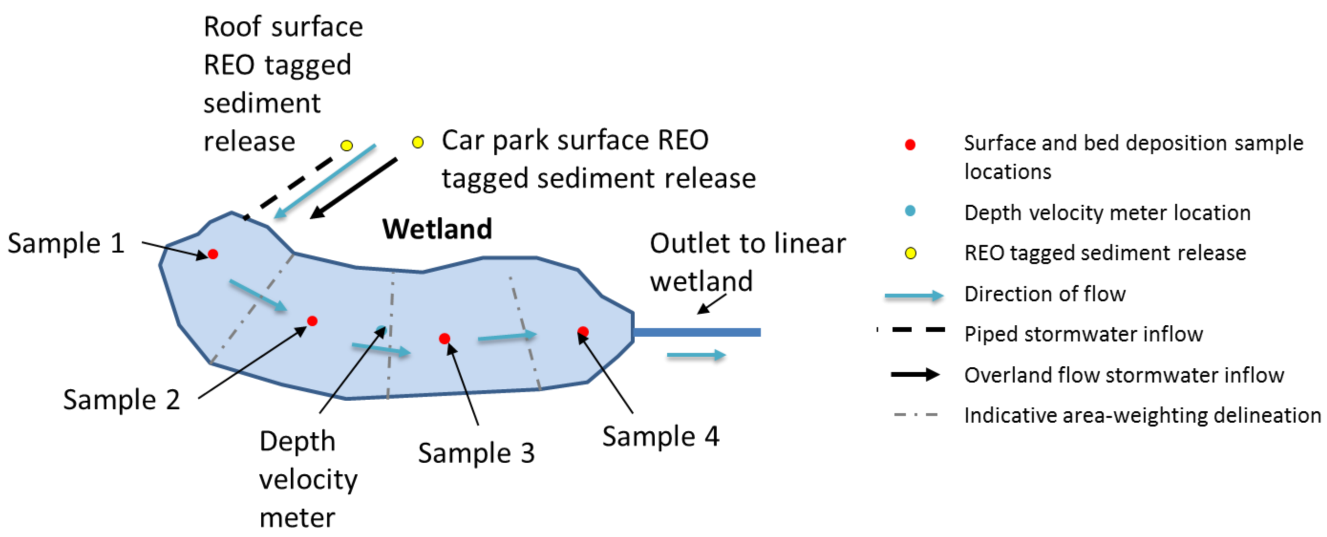


Figure 5.2 Schematic of wetland sample locations

5.3.1 Sediment detention efficiency

Tagged sediment was monitored in the wetland over 52 weeks (Networks 1 and 2). Using simple mass balance analysis based of representative area-weighted suspended and bed deposition sample results (indicative area-weighing is illustrated in Figure 5.2 and Appendix V), the total tagged sediment in the wetland was quantified. Analysis of the percentage of total tagged sediment leaving the wetland defined the wetland sediment detention efficiency over the monitoring period. Figure 5.3 illustrates the wetland detention efficiency of tagged fine sediment from surface (Figure 5.3a) and sub-surface (Figure 5.3b) supply for the first of the four REO tagged sediment releases.

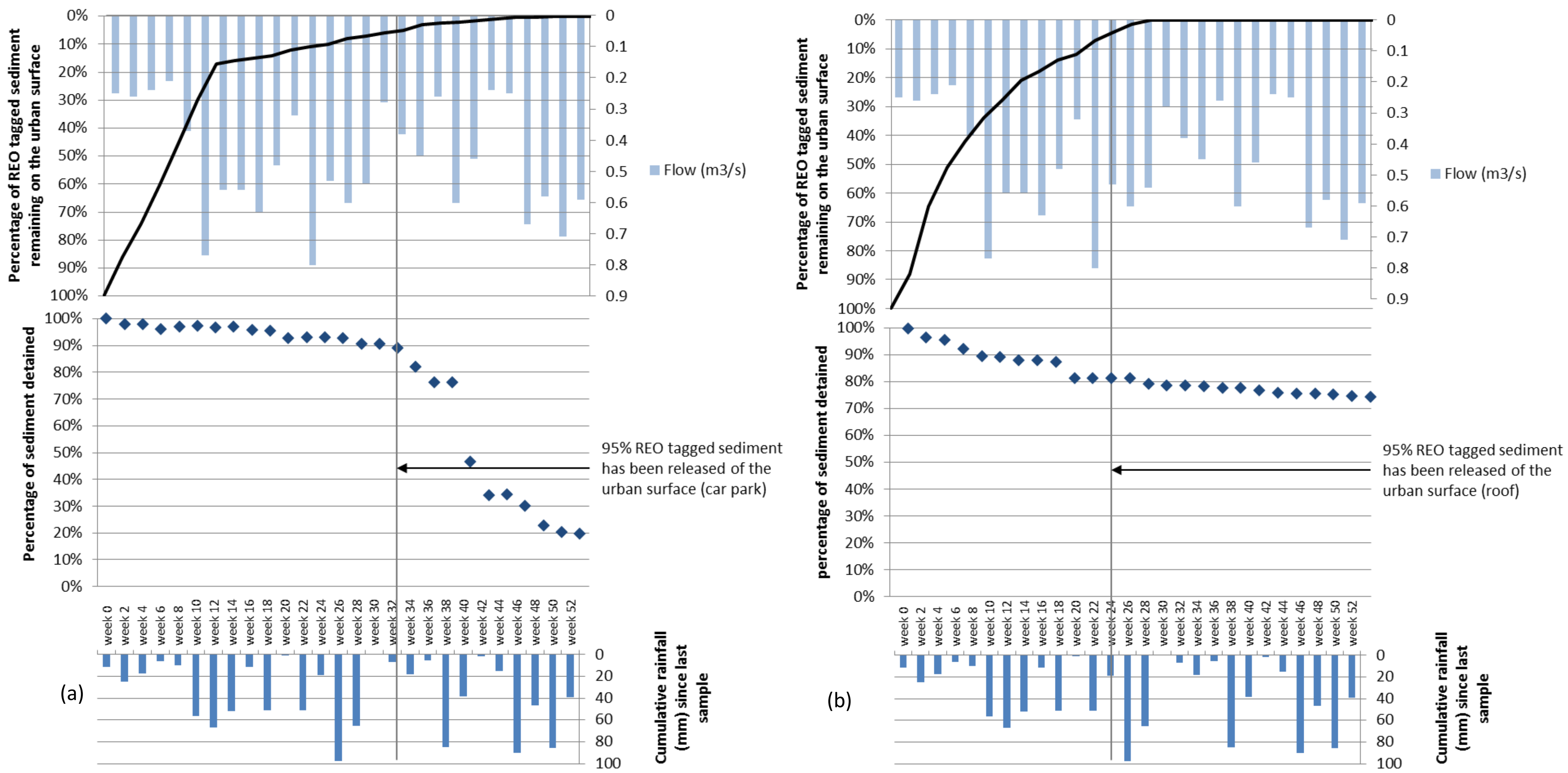


Figure 5.3 Wetland sediment detention efficiency (%) for REO tag time-stamped Release 1 for (a) surface supply of Network 1 and (b) sub-surface (piped) supply of Network 2. Percentage data are relative to the amount of REO tag initially released at the inlet (100%). The amount detained within the asset is the summation of the sampled REO amounts (suspended or bedload) within the wetland area corrected as representative of the sample location. The notable decrease in Figure 5.3a at week 40 is potentially due to the high number of events (19) and low ADD during the sample period (144hrs) and prior to sampling (1.5hrs) resulting in greater sediment movement within and from the wetland.

Three important findings arise from Figure 5.2:

Firstly, although ~100% of sediment is detained after the initial stormwater inflow (week 0), the % detained progressively decreases with subsequent runoff events. Over the first year of sediment transport, this wetland sediment detention decreases from 95% down to 75-20% depending on the supply, rainfall and runoff conditions and flow path design. This indicates re-entrainment of tagged fractions for redistribution within and discharged from the wetland asset.

Secondly, the absolute values of detention efficiency indicates that a single wetland asset does not necessarily achieve SuDS design guidance (75-90% detention) of detention over the longer term. Rather, the current 90% UK reported efficiency (Woods Ballard et al. 2015) is achieved for ~ 6 months (90.5% at week 30 in Network 1, 89.3% at week 8 in Network 2) after tagged sediment provision/release. Importantly, Figure 5.3a (supported by Chapter 4, Section 4.2.2) suggests that the overland supply of tagged sediment to the asset is a slow-release process; it is this supply which maintains the high percentage of detention. Thus, once supply is exhausted (>95% of the surface released material has entered the wetland (week 32 for Network 1) the surface supplied wetland response is a dramatic and rapid decrease in detention performance more reflective of general asset performance (rather than supply complexities). This is not mirrored in the sub-surface supply of Network 2, where supply is more instantaneous and trends in Figure 5.3b are more representative of general asset performance.

Thirdly, the detention efficiency is potentially related to the runoff routing, with sub-surface inflow paths of Network 2 (Figure 5.2b - pipe) resulting in stronger wetland sediment detention (≥74%; Figure 5.3b) compared to surface overland inflow (≥20%; Figure 5.3a). This may imply that entry of tagged sediment into the surface waters (Figure 5.3a) results in retention of these fine particles suspended in the water column, hence subjecting this tagged sediment to faster routing and flushing from the asset, or longer hydraulic residency requirements for sediment entering the asset via surface (overland) due to the greater settling depth and therefore the long settling time requirements.

Given that the findings above were specific to a given flow sequence, the tagged sediment experiment was repeated a further three times to ascertain sensitivity to runoff sequence. Releases occurred 12 weeks apart and Figure 5.4 presents the experiment results.

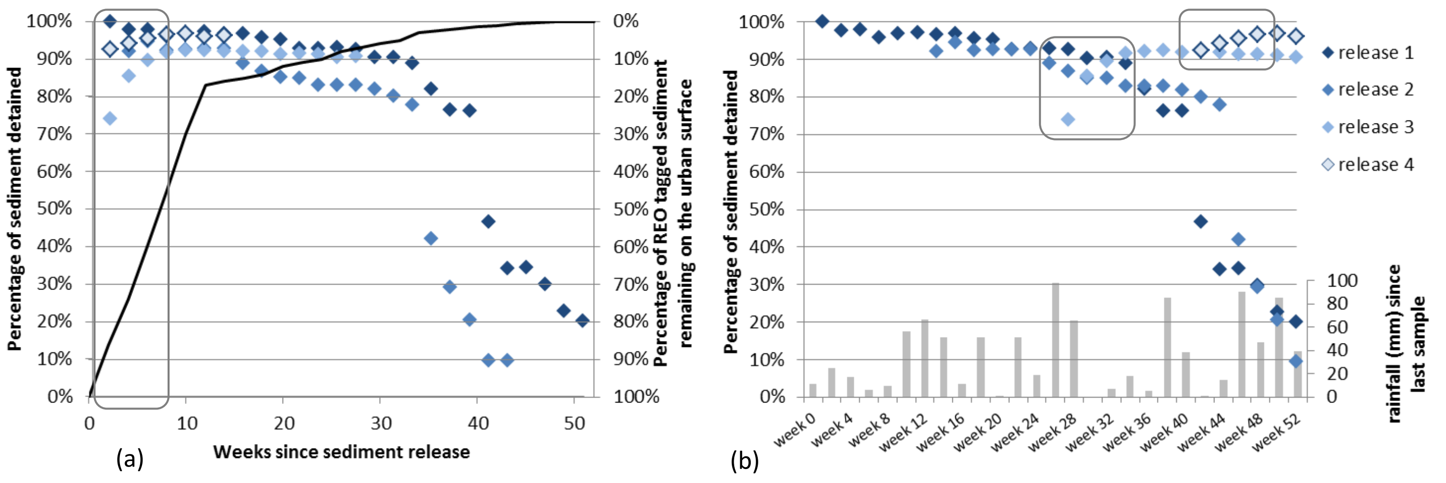


Figure 5.4 Network 1 Wetland. Figure 5.4 (a) presents the REO tagged sediment detained within the wetland (as a %) from the time of tagged sediment release and the corresponding surface release (percentage of REO tagged sediment remaining on the urban surface). Figure 5.4 (b) illustrates the REO tagged sediment detained in Network 1 wetland relative to the commencement of sampling (time stamped, weeks since the first sample) and the corresponding fortnightly cumulative rainfall (mm).

The tagged sediment detention over first 8 weeks illustrated a decline in Release 1 and 2 but a rise in Release 3 and 4 (until week 8). This suggests that Release 3 and 4 were more directly influenced by initial supply (the release of ~80% tagged sediment off the car park surface (Figure 5.4a)), while Release 1 and 2 may respond more strongly to rainfall (Figure 5.4b). The initial increase (kick, highlighted in Figure 5.4) in detention in Release 3 and 4 may alternatively or additionally occur due to high rainfall occurrence during the initial release. The increase in detention efficiency is noted at the beginning of each affected Release monitoring period, not across all releases at one time. This may be explained by cohesive sediment binding adhering floc particulates to the urban surface, easily removed sediment being previously washed off, hiding or burial of fine sediment on the urban surface and within the wetland resulting in the most recently released sediment *only* being notably affected by the rainfall-runoff. Figure 5.4 results from Network 1 (overland supply) Release 1, 2, 3 and 4 show a similar trend and temporary detention percentage for up to the first 26 weeks (6 months). Overall, this demonstrates detention efficiency ≥ 80%, but with a progressively decreasing trend for this initial period of multiple events. This provides support and validation to the initial release (Release 1) results shown in Figure 5.3 and efficiency of asset (highly effective ≥80%, Chapter 4.3.3). Only Release 1 illustrates sediment detention efficiency results higher than 80% for longer than 26 weeks, indicating a significant decrease in detention to 20% after 52 weeks. This is in line with description and explanation presented for

Figure 5.3. Figure 5.4 also shows data for Release 2 which follows the long term trends of Release 1 for decreasing detention over multiple events towards a constant long term (≥ 6 months) detention of >20%. However, for Release 2 the detention loss is experienced earlier in the monitoring period (decline below 80% occurs at week 16); as the particle size distribution, release methodology and receiving network were not changed between releases it is proposed that the overland supply of Release 2 to the asset was faster and more likely *en mass* than for other more progressive releases. When considered by sampling date rather than period since tagged sediment release (Figure 5.4(b)) Release 2 decline follows that of Release 1, suggesting that rainfall-runoff characteristics over this period may be the key influence.

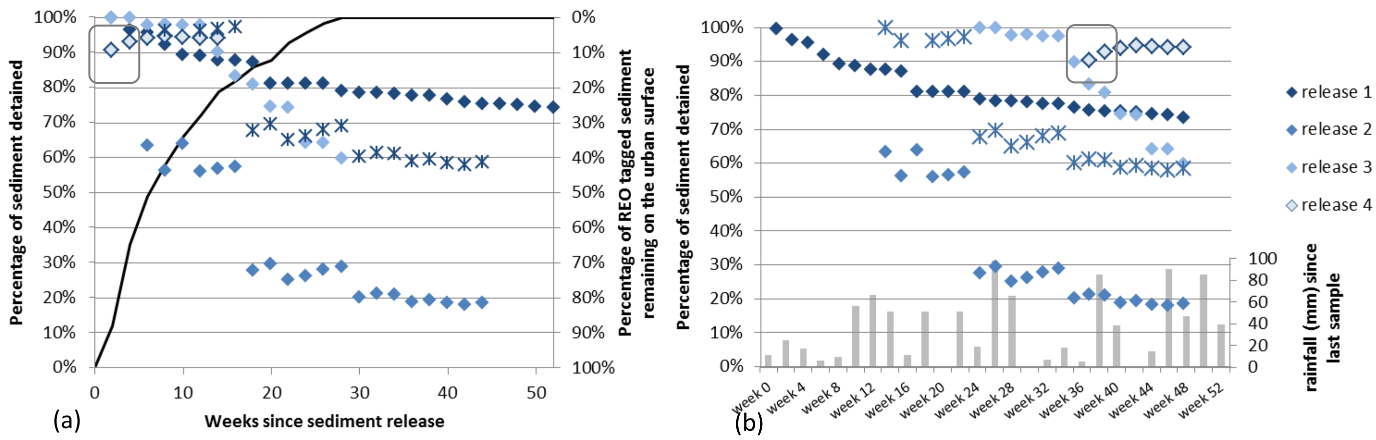


Figure 5.5 Network 2 Wetland. Figure 5.4 (a) presents the REO tagged sediment detained within the wetland (as a %) from the time of tagged sediment release and the corresponding surface release (percentage of REO tagged sediment remaining on the urban surface). The star points (*) in Figure 5.5 illustrate Release 2 results if the initial ~70% decrease has not occurred. Figure 5.5 (b) illustrated the REO tagged sediment detained in Network 2 wetland relative to the commencement of sampling (time stamped, weeks since the first sample) and the corresponding fortnightly cumulative rainfall (mm).

Network 2 Release 1, 2 and 4 indicate similar sediment detention up to 16 weeks after release, with all tagged sediment demonstrating detention ≥ 80% (illustrated in Figure 5.5(a)). Release 4 illustrates a slight increase in the initial dataset suggesting potentially high rainfall conveying fine, easily released and suspended sediment through the system with detention efficiency increasing and the release matures and cohesive material potentially aggregates and becomes deposited. Release 2 show a low initial sediment detention efficiency, >70% in weeks 0-8. This may be due to a notably lower than average dry period directly after tagged sediment release, 96 hours of ADD over the preceding fortnight and 0 hours at the time (week 14) compared to monitoring period average ADD of 206 and 21 hours respectively (Chapter 4, Table 4.1) resulting in greater urban surface wash-off potential during this period. This significant (~35% of the total tagged sediment) initial drop in sediment detention may result from modified urban surface (roof area) management; such as flushing of the roof down pipes or cleaning of the roof gutters. This could result in a significant flush of tagged sediment into and potentially through the wetland in suspension. The purple star points illustrated in Figure 5.5a represent Release 2 results without the initial ~70% decrease. When Release 2 detention efficiencies are 'normalised' (initial detention rate rectified to ~>90%) the detention trend is similar to that of Release 1, 3 and 4. Release 2 and 3 show a sharp decline during week 38, potentially in response to rainfall (>80mm in the fortnight prior to sampling (Figure 5.5(b)). A similar sharp decline is illustrated in Release 2, week 26 (Figure 5.5(b)) in correlation with a high rainfall occurrence. The sharp declines presented in Releases 2 and 3 are not found in Release 1. This may be due to hiding or burial[1]of Release 1 sediments below that of Release 2 and 3, thus resulting in preferential local re-suspension of more recently released tagged sediment. However it may also illustrate the influence of preferential conveyance of finer sediment; the influence of rainfall in weeks 26 and 38 may show sharper sediment detention efficiency declines in Release 2 and 3 due to a greater quantity of fine sediment availability, whereas Release 1 fine sediment may have already been conveyed downstream or become flocculated/aggregated (cohesive influence) thus resulting is a much smaller sediment detention efficiency decline.

At first glance, Release 1 and 2 data appear distinctly different, however both datasets show a slow but consistent sediment detention efficiency decline after week 16 (Figure 5.5(a)). The slower rate of sediment detention efficiency decline within Release 1 and latterly within Release 2 may occur due to the maturity of the deposits, increasing the potential of sediment burial within the wetland and therefore the slower rate of decline in detention. Thus, if the initial significant sediment loss is Release 2 is temporarily disregarded, Release 1 and 2 illustrate comparable decline (trend) in detention from week 16 onwards (purple stars in Figure 5.5(a). Thus Release 2 presents a sediment detention efficiency trend that compares, at different periods, to both Release 1 and 3,

[1] Particle hiding occurs when larger particles or objects on the bed shelter or temporarily detail finer sediment within the upstream low pressure eddy zone. Burial of sediment is when a particle resting on the bed is covered by more recently deposited material, resulting in the original particle becoming too deep in the bed to become easily re-entrained (Ferguson et al. 1996).

comparative to rainfall and illustrating the slower decline potentially occurring due to longer term deposition and possible burial.

In addition, Figure 5.5 can be directly compared to Figure 5.4 to ascertain distinction in overland versus subsurface piped supply, given that all releases were synchronized temporally and specific to the environmental controls of the same J4M8 site. Crucially, the results presented confirm that fine sediment moves through the wetland over multiple events and that sediment detention declines over multiple events. Generally, detention efficiency is high (≥ 90%) for 8 weeks in a wetland system, independent of overland or subsurface supply mode, and falls within design expectations or report values (≥80%) for ~15 weeks (4 months). The rate of this decline is shown to vary between experiment repetitions and networks, and therefore the variance and standard error of the decreasing sediment detention trends has been calculated and presented in Table 5.1. However, two key points are concluded from the Figures 5.4 and 5.5: (i) that for multiple event timeframes in excess of these data, the detention rate decreases towards 20% detention efficiency of the asset over the longer term of up to a year; (ii) data suggest an event-related (individual or cumulative event) trigger for a switch from progressive loss of detention to a more rapid rate of decline in detention efficiency, which is considered in further detail in Chapter 6.

Table 5.1 SuDS asset sediment detention efficiency (%)

Monitoring period	release	week 2	week 8	week 16	week 24	week 32	week 40	week 48	week 52
Wetland – network 1									
average	0	92%	95%	91%	89%	83%	28%	23%	19%
variance		4%	2%	4%	4%	5%	19%	0%	0%
StdError		1%	1%	1%	1%	1%	5%	0%	0%
Wetland – network 2									
average	0	96%	86%	65%	58%	49%	47%	75%	73%
variance		3%	13%	27%	22%	29%	29%	0%	0%
StdError		1%	3%	*7%*	*6%*	*7%*	*7%*	0%	0%
Wetland – average									
average	0	94%	90%	78%	73%	66%	37%	49%	46%
variance		4%	10%	23%	22%	27%	26%	26%	27%
StdError		2%	4%	*8%*	*8%*	*9%*	*9%*	*9%*	*10%*

The variance across the four release results was calculated by direct comparison of detention efficiency, irrespective of rainfall occurrence and flow event characterization over the relevant period. The aim of the direct comparison was to determine the general overall variance in trend, rather than specific influences of variations in the trends (discussed in Chapter 6). Analysis of the variance between datasets, through calculation of standard deviation for the total sample dataset, highlighted the consistency of the trends represented in Figures 5.3 and 5.4. There is lower variability between release results and lower standard error in Network 1, supply by overland flow. Monitoring results from sub-surface (piped) supply (Network 2) are shown to be more variable with an elevated standard error of the mean (an average shift from 2% up to 4%, with a maximum standard error of 7%). This elevated variance is due to the significant initial decrease in sediment deposition efficiency of Release 2. If this initial drop (~35%) is not included in the dataset (i.e. the entire trend is raised 35%, such that the rate of decline is unchanged but the initial detention efficiency is >90% in line with Release 1, 3 and 4) then variance in Network 2 becomes similar to Network 1 (variance: 2-8%; standard error: 1-3%). All results were calculated to have an acceptable (below 10%) standard error, supporting the calculated average sediment detention for the monitored Network wetlands. If an overall average, taking into consideration the results from both networks, is created (Wetland-average in Table 5.1) then an overall trend can be seen between the maximum and minimum recorded sediment detention efficiencies for the wetland (presented in Figure 5.6).

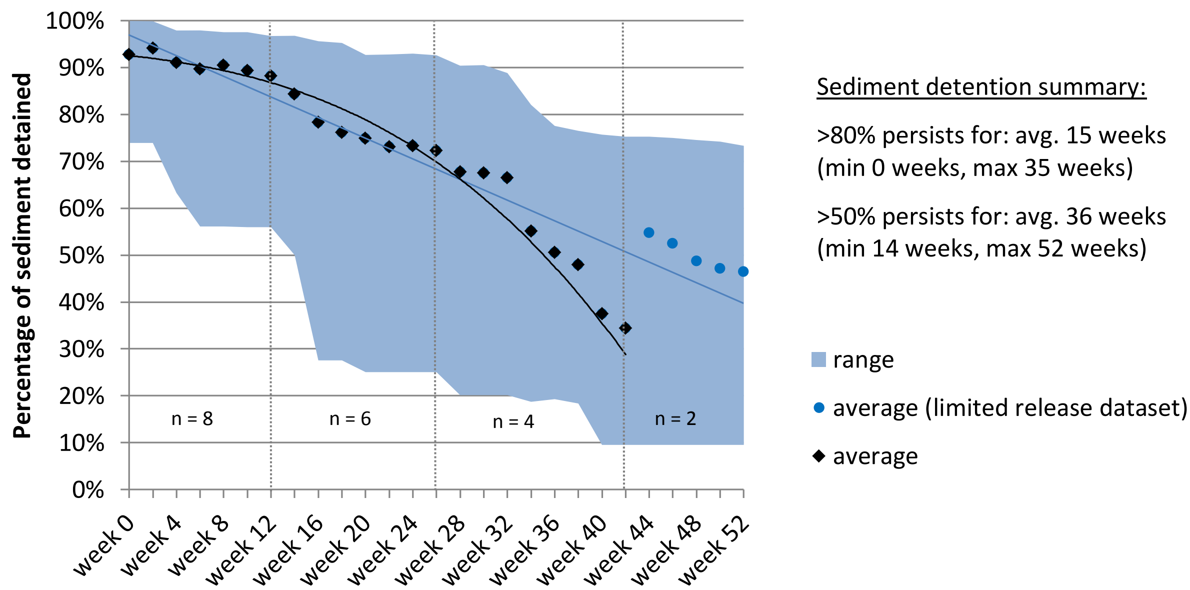


Figure 5.6 Average wetland sediment detention trend. The black points and trend line indicate the average wetland sediment detention efficiency trend determined from multiple release results (Release 1-4). The blue points and trend line illustrate the final 10 weeks of Release 1 (the longest monitored REO tagged sediment release) and do not have the equivalent sampling replication as the black data points (n=2 for blue points, n ≥ 4 for black points). The surface and

sub-surface results have been combined to create Figure 5.6 as many wetland SuDS assets incorporate both surface runoff collection and piped stormwater discharge inflow.

Experiment repetition extended over 42 weeks of the monitoring period (after week 12 there are more than 2 results datasets – more than 1 results dataset for each Network). The overall average wetland sediment detention efficiency trend is presented in Figure 5.6 (trend lines), within the shaded graphical extent of the maximum and minimum recorded sediment detention results. The black trend line and points have greater than 2 degrees of freedom and are therefore considered to be more statistically representative of the wetland trend. If overall average trend is considered (black trend line), calculated as the average for each sampled fortnight for all releases across both Network 1 and 2, a gradual but continual declining trend in sediment detention efficiency is illustrated. This emphasizes, and is mirrored by, the general overall shift in sediment detention efficiency range (the shaded blue range presented in Figure 5.6). This analysis is focused on the monitored wetland in Networks 1 and 2, but provides key insight into the potential sediment detention capability of SuDS wetlands in general.

5.3.2 *Sediment deposition locale*

Further analysis of the wetland sediment dataset was undertaken to identify the location of sediment detention within the wetland. Figure 5.2 illustrates the sample locations across the wetland for Networks 1 and 2. Sediment was found to deposit across the length of the wetland flow path. However, the preferential deposition location is illustrated in Figure 5.7.

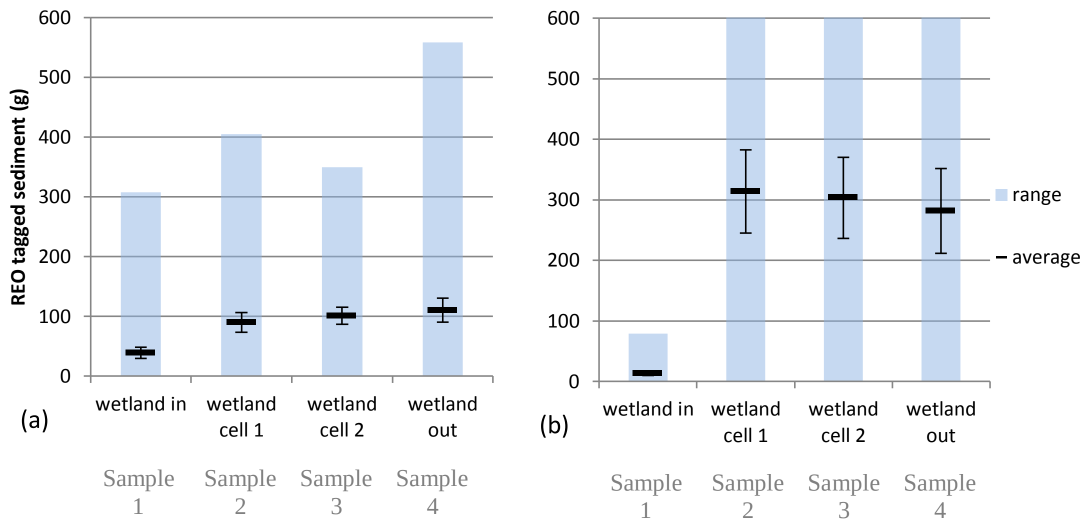


Figure 5.7 Network 1 (a) and Network 2 (b) Wetland preferential deposition locations. REO tagged sediment results are representative of sample period deposition across the sample

specific SuDS area. The maximum range for Network 2 (Figure 5.7b) extend to 2.1kg (cell 1), 2.0kg (cell 2) and 1.9kg (wetland out) but the vertical axis has been truncated to visually present the average and standard deviation of the deposition results. Wetland sample locations, presented in Figure 5.2, are noted in grey to provide further graphical context.

The lowest sediment deposition mass was retrieved from the wetland inlet for both networks. This is most likely a facet of sampling from within a zone of significant fluid mixing where turbulent shear stresses are sufficiently high so as to keep fine tagged material within suspension, thus precluding deposition into the bed samplers. The localized piped inflow of Network 2 may lead to stronger near-bed shear stresses in this bed sampler location (hence higher suspension and lower bed deposition) than the more spatially distributed surface sheet flow input of Network 1 where natural settlement of fractions may lead to the higher recorded mass. Cell 1 (sample 2 in Figure 5.2) of the wetland shows the highest sediment deposition results for sub-surface supplied sediment laden stormwater. Cells 1, 2 and the wetland outlet performed comparably in terms of detention, suggesting that within the wetland shear stress was relatively uniform once away from the inlet. REO tagged sediment detention across all three cells occurred with deposition of around 300g per monitoring period (fortnight). Sediment arriving at the wetland via overland flow (Network 1, Figure 5.7(a)) showed considerably lower overall deposition, ranging from 38 to 110g per monitoring period. The lower deposition values are due to the difference in supply path (overland vs piped sub-surface flow) and in supply rate (sediment wash-off from the respective urban surfaces, Chapter 4.2.2). The higher deposition in cells 1, 2 and outlet may also be due to flocculation of cohesive material, occurring along the wetland flow path and resulting in greater fine sediment material settling within the latter wetland extent. Results from Network 1, cell 1, 2 and the outlet cell, are also relatively consistent at a deposition rate of approximately 100g per monitoring period. The supply flow path therefore may have an influence on the deposition rate within the network, seen through the difference between Network 1 and Network 2 wetland deposition values. This analysis also shows that it is important to include multiple cells within a wetland design as the inlet cell is important with regards to delivery and re-entrainment while cells 1, 2 and the outlet are efficient in the detention of fine sediment.

5.3.3 Shift in particle size distribution

The deposition of sediment within the wetland results in a shift in the particle size distribution of REO tagged sediment in the wetland outflow when compared to the inflow. Analysis of the surface and bed sediment trap sample particle size distributions illustrates that the particle size influence of wetland sediment detention on surface flow and bed deposition results, illustrated in Figure 5.8.

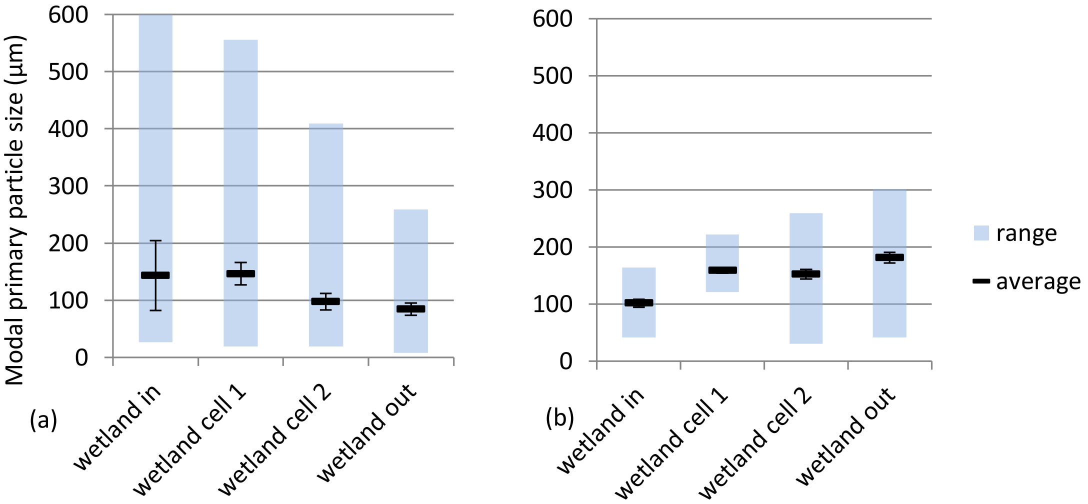


Figure 5.8 Modal particle size for wetland suspended sediment (a) and bed deposition (b) sediment samples for the wetland (Network 1 and 2). The modal particle size range for the 'wetland in' sample area extends to 1859µm; the vertical axis has been truncated at 600µm to provide visibility of average and standard deviation result details.

Particle size distribution analysis of the wetland sediment samples identified that samples were generally monodisperse (unimodal), with a well-defined modal particle size range found within each sample. As would be expected, the surface particle size distribution shift is inverse to the bed deposition particle size distribution shift. This illustrates that the sediment detailed temporarily on the wetland bed is subsequently removed from the sediment mass in suspension, proving a direct interaction between surface and bed deposition sediment masses. Figure 5.8 illustrates that the suspended sediment is most coarse at the inlet and finest towards the outlet. This may be due to the natural sediment settling processes[2], with larger material settling out of suspension through the wetland flow path thus resulting in a smaller modal particle size (and particle size distribution).

The bed deposition size is found to be finest at the inlet and between 151-181 µm across the remainder of the wetland. The fine sediment deposition found at the wetland inlet

[2] Sediment settling processes: the process by which the density of the sediment and velocity flow of the fluid influence the sedimentation or settling velocity of a particle (Baldock et al. 2004). Slower fluid velocity and heavier sediment particle achieve faster settling velocity and therefore greater sediment deposition of this particle size sediment. Cohesive sediment (i.e. clay), when forming a cohesive particulate mass, can achieve greater settling velocities and therefore deposition than individual sediment particles, thus falling out of suspension and becoming deposited on the flow path bed (Berlamont et al. 1993).

may correlate to the lower deposition within this area (Figure 5.7) and the turbulent flow supportive of larger sediment (re-)entrainment and suspension. The suspended sediment particle size range reduces from >600 to 250 µm from the inlet to the outlet, indicating that near inlet turbulence and shear stress is dispersed with distance/volume. The bed deposition maximum particle size increases up to 300 µm. This suggests that the fraction sub to 30 µm are mobile and subject to deposition - re-entrainment within and through the wetland asset.

In general, the wetland monitored in Networks 1 and 2 detained a greater quantity of larger (>100µm) sediment particles than smaller ones, effectively shifting the modal particle size and particle size distribution down (towards 0.45µm). The wetland functions to decrease the overall quantity of larger sediment conveyed downstream, with a lesser impact on the conveyance of fine (clay sized) particles.

5.3.4 Wetland conclusions

A summary of the wetland sediment conveyance and detention efficiency research findings is provided. The key research findings reflecting wetland fine sediment transport discussed in this section of Chapter 5 are summarised below.

Fine sediment moves through the monitored wetland over multiple rainfall-runoff events. It is not illustrated (in the monitored field study SuDS) to be permanently detained once the supply event ceases. While there is notable variance in monitored results, all data show a decreasing sediment detention efficiency over multiple events.

The initial temporary detention efficiency of the wetland, during week 0-2, was found to be high, ≥90% of the supply. The wetland was found to achieve >80% sediment detention efficiency for 15 weeks (average).

After the monitored period of 52 weeks, the fine sediment detention efficiency of the wetland was found to fall between 19% and 73% of the initial inflow. The exhaustion of sediment supply is found to influence the detention efficiency decline.

Sediment detention occurs across all cells of the wetland, with notable limited detention close to the inlet. The inlet design appears to bear some relation to the sediment detention efficiency (stormwater runoff entering via sub-surface pipe vs overland flow). Incorporation of multiple cells (inlet plus at minimum 1 cell) in a wetland design helps achieve the maximum wetland potential detention efficiency, with wetland cell 1 to outlet illustrating consistently greater detention mass (g) than found at the inlet.

Suspended sediment modal particle size decreases through the wetland, resulting in the removal of coarser sediment from suspension. Bed deposition shows limited variation within the wetland (cells 1-outlet) with an increasing particle size range moving downstream through the wetland.

5.4 Linear Wetland

Tagged sediment from both Network 1 and 2 passed from the wetland into and through the monitored linear wetland (sample locations indicated in Figure 5.9, J4M8 network outline illustration in Chapter 3, Figure 3.2, additional indicative area-weighing information is provided in Appendix V). All sediment laden flow entering the linear wetland, in Networks 1 and 2, was conveyed via the wetland outlet. The inlet into the linear wetland is a natural channel cross section (trapezoidal channel with 0.5m bed width, 1m depth and 1:1 bank slope) with soil/sediment substrate, vegetated bed and banks. The inflow from Network 1 and 2 follow the same flow path and are influenced by the same rainfall-runoff conditions.

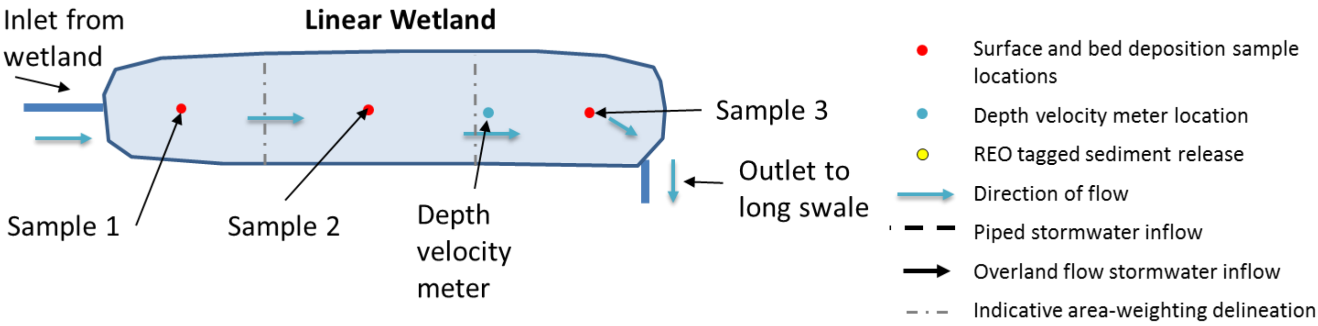


Figure 5.9 Schematic of linear wetland sample locations

5.4.1 Sediment detention efficiency

Although temporary detention of REO tagged material occurred within the wetland, Section 5.4 shows material it to discharge from the wetland, thus providing an unsteady supply to the linear wetland. The inflow supply rate of tagged sediment into the linear wetland was assumed equal to the measured discharge rates from the upstream wetland (Figure 5.4 and 5.5); this is justified as the channel connection for the assets was of short distance (less than 1m) and of higher flow rate (and shear stress) than that of the wetland outlet such that tags entrained would remain in transport (suspension or bed load). A mass balance analysis of the quantity entering and leaving the linear wetland asset was therefore possible and the results (in percentage detained within the linear wetland) are graphically presented in Figure 5.10.

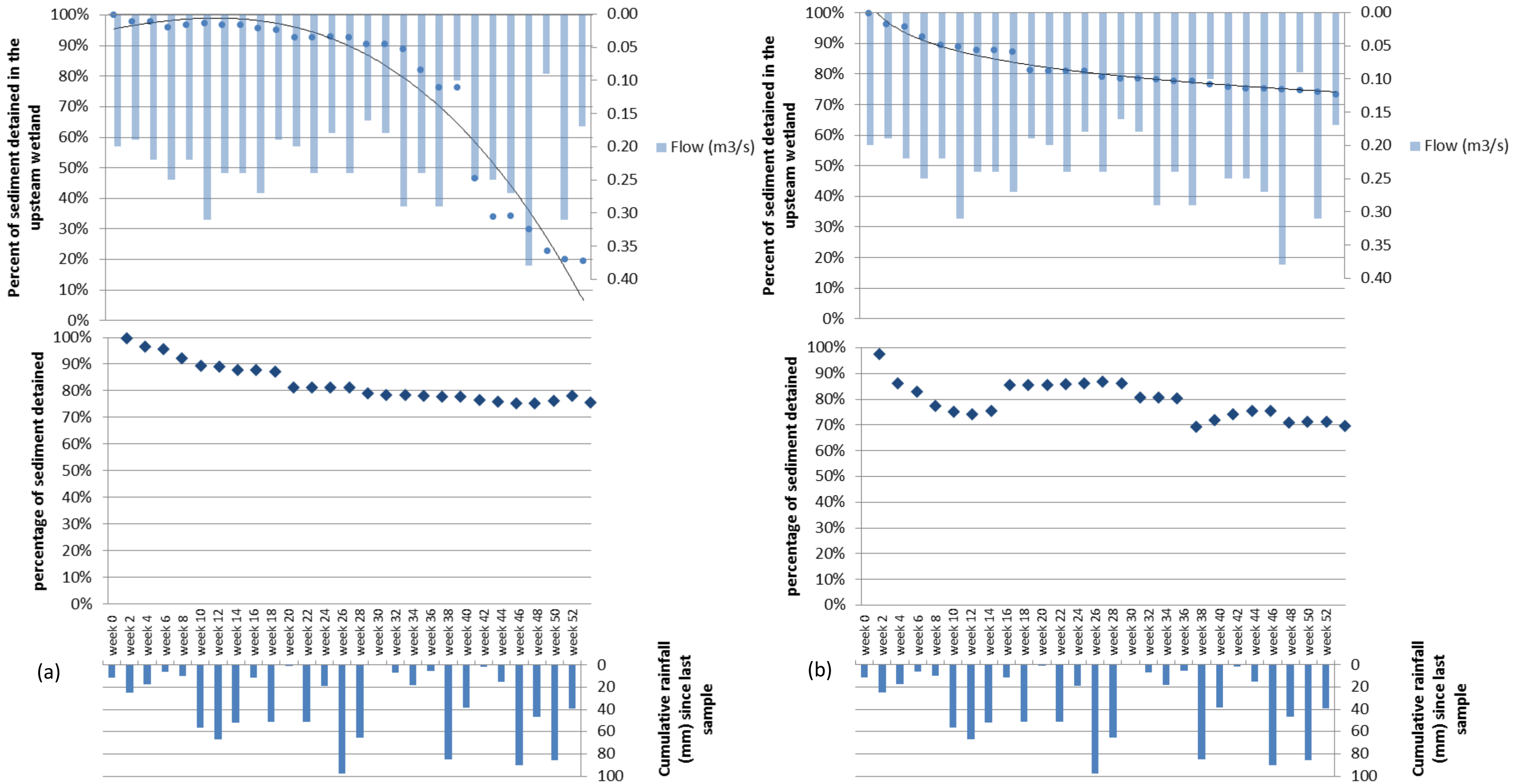


Figure 5.10 Linear wetland sediment detention efficiency (%) for Release 1 for (a) Network 1 and (b) Network 2. Percentage data are relative to the amount of REO tag initially released at the inlet (100%). The amount detained within the asset is the summation of the sampled REO amounts (suspended or bedload trap) within the wetland area corrected as representative of the sample location

Initial sediment detention (for the first 6-8 weeks after the tagged sediment release) within the linear wetland appears higher than the longer term trend, functioning highly efficiently at >80% (Chapter 4.4.3). Between Weeks 6 and 8, after multiple rainfall-runoff events, the linear wetland fine sediment detention efficiency was found to drop to a long term efficiency varying between 75-89%. A generalization of data from Figure 5.10 is that Network 1 performs ~80% detention variability at least +/-10%, whilst Network 2 performs more consistently at ~70%. Network 2 is more notably supply limited (there is a greater average sediment detention efficiency within the upstream wetland of Network 2 than Network 1, resulting in more limited tagged sediment entering the linear wetland in Network 1 due to the conveyance of tagged sediment through the upstream wetland). The supply limitation may influence the linear wetland detention efficiency, with greater tagged sediment supply resulting in lower mass detention within the linear network (i.e. Network 2 results).

As described in Section 5.3, tagged sediment was released onto the supply surfaces four times, with the results presented in Figure 5.11 across the full 52 week monitoring period.

The multiple release results for the linear wetland in both Networks 1 and 2 show a general sediment detention efficiency of between 45-95% (with the exception of Release 3 in Network 2). For Network 1, Release 1 and 2 show similar trends in decline over the first 40 weeks of monitoring (Figure 5.11a). Release 2, 3 and 4 all show a declining trend in sediment detention efficiency to less than 55%, and 20% in Release 3. Release 3 and 4 may illustrate a strong response to rainfall-runoff conditions (Figure 5.11(b)); week 46-52 show rainfall (depth mm) greater than average fortnightly rainfall . Release 1 and 2 present a slower rate of sediment detention efficiency decline; potentially in response to the delay in wetland sediment detention efficiency decline (Figure 5.11 (a)). The results suggest that Network 1 shows a general decline in sediment detention efficiency and that the sediment detention efficiency levels achieved over an extended monitoring period are generally below 90% (below 80% after week 26).

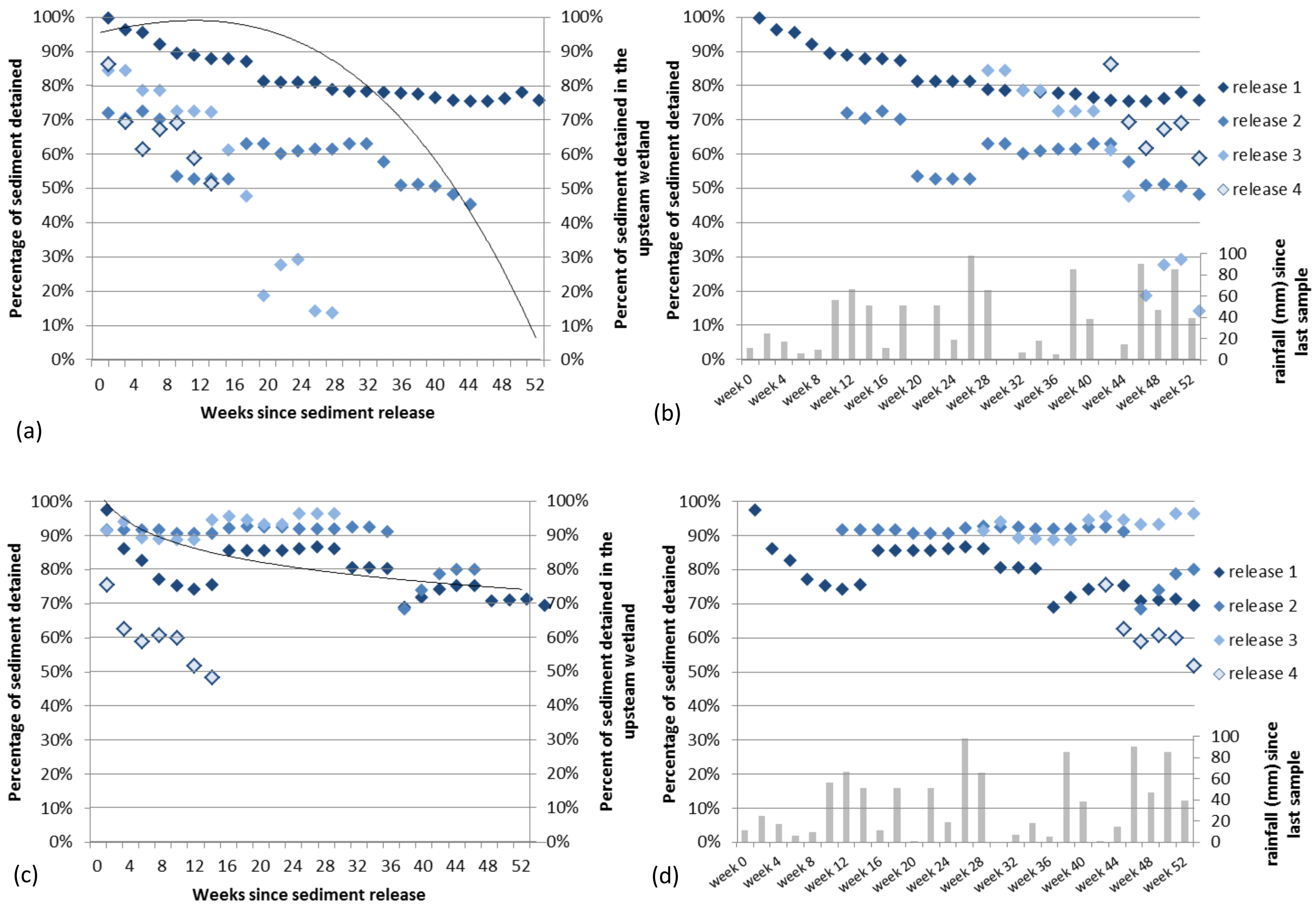


Figure 5.11 Tagged fine sediment detention through the linear wetland for Network 1 (a, b) and Network 2 (c, d). Figure 5.11 (a and c) present the REO tagged sediment detained within the wetland (as a %) from the time of tagged sediment release and the corresponding surface release (percentage of REO tagged sediment remaining on the urban surface). Figure 5.11 (b and d) illustrated the REO tagged sediment detained (in Network 1 and 2 respectively) linear wetland relative to the commencement of sampling (time stamped, weeks since the first sample) and the corresponding fortnightly cumulative rainfall (mm).

Network 2 results show, generally, more constant and higher detention efficiency than Network 1 data. This more constant and elevated detention level may result from the higher upstream (wetland) sediment detention and continuous upstream slow but steady sediment release into the linear wetland (with exception to Release 2). Release 2 and 3 also show excellent performance at ~90% for the first 30 weeks, before detention falls to <80% (Release 2) likely due to rainfall-runoff events (similar trend dip in sediment detention efficiency is visible in Release 1 (Figure 5.11(a)) and 4 (Figure 5.11 (d)) but at a lesser magnitude. Network 2 Release 4 is distinct in Network 2 trends progressively falling to <50% detention within the 12 weeks monitored; this mirrors the sediment detention trends found in Network 1Release 4, suggesting two important points: firstly that there is some similarity in the function of the linear wetlands and, secondly, that

Release 4 may have been more strongly influenced by rainfall (more numerous events and greater rainfall depth) during or immediately after the tag Release. The supply of sediment by the wetland is considered to be the primary cause of the difference between stabilization in Network 1 and 2.

The variation in repetitive release results and the overall detention efficiency trends for the linear wetland within Network 1 and 2 are defined in Table 5.2.

Table 5.2 SuDS asset sediment detention efficiency (%)

Monitoring period	release	week 2	week 8	week 16	week 24	week 32	week 40	week 48	week 52
Linear wetland - Network 1									
average	0	80%	71%	66%	52%	67%	61%	78%	75%
variance		12%	13%	16%	28%	10%	14%	0%	0%
StError		3%	3%	4%	7%	3%	3%	0%	0%
Linear wetland - Network 2									
average	0	84%	79%	91%	92%	86%	76%	71%	64%
variance		12%	12%	4%	4%	5%	2%	0%	0%
StError		2%	3%	1%	1%	1%	1%	0%	0%
Linear wetland - average									
average	0	82%	75%	78%	72%	77%	70%	75%	70%
variance		9%	13%	17%	28%	12%	13%	3%	5%
StError		3%	5%	6%	10%	4%	5%	1%	2%

When the Network datasets are averaged, to find a general overall trend in detention efficiency, the standard error of the mean is shown to be small (<10%). The overall average trend for Network 1 shows a shallow decrease of 19% over the first 40 weeks of monitoring. The standard error for Network 1 average values is small (<10%), illustrating confidence in this finding. From week 42-52 there is only one release dataset, without replication, and therefore there is no variance or standard error of the mean results for these final weeks. The average overall results for Network 2 show a similar trend over weeks 0-40. As such, the overall trend can be considered as relatively linear and with a shallow (8%) average decrease in sediment detention over the 40 week period of replicated sampling. The variance within the network specific datasets (average of N1:12%, N2:5%) are smaller than for the average linear wetland (13%), and the upstream wetland (N1: 5%, N2:15%, wetland-average: 21%) (Table 5.1 and 5.2). The average and extent of sediment detention efficiency results for the linear wetland are presented in Figure 5.12.

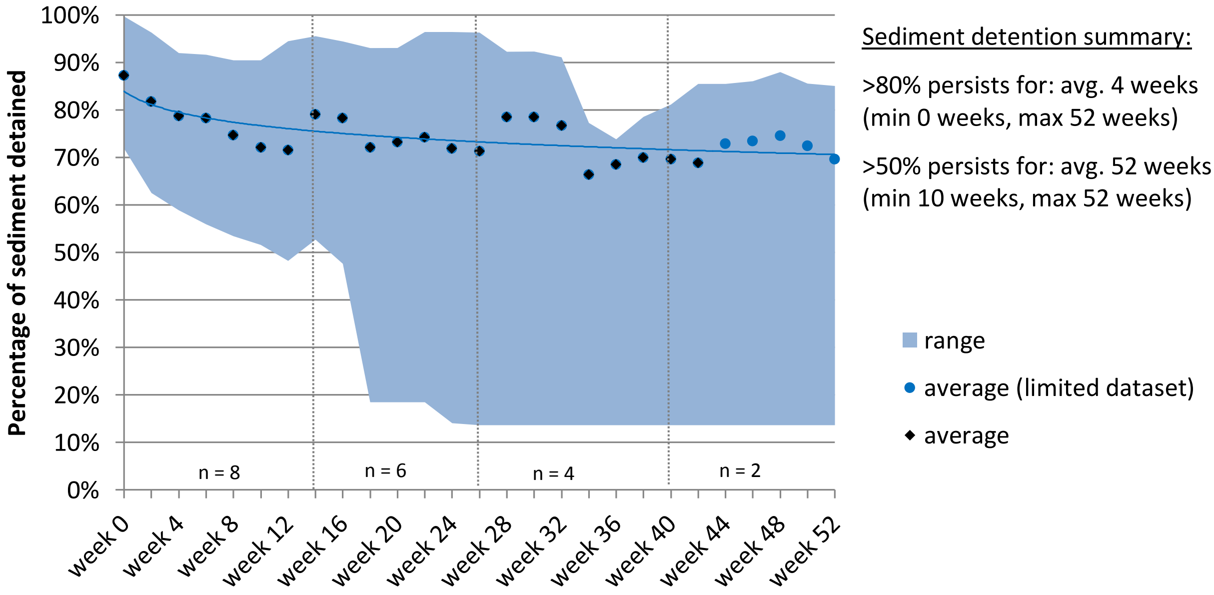


Figure 5.12 Average linear wetland sediment detention trend. The black points and trend line indicate the average wetland sediment detention efficiency trend determined from multiple release results (Release 1-4). The blue points and trend line illustrate the final 10 weeks of Release 1 (the longest monitored REO tagged sediment release) and do not have the equivalent sampling replication as the black data points (n = 2 for blue points, n ≥ 4 for black points).

The linear wetland within Network 1 and Network 2 both illustrate a slow (with exception of N1, R2) decline in sediment detention over the monitoring period. When the detention efficiencies from both Network 1 and Network 2 are considered for this linear wetland, an overall trend (without acknowledgement of the upstream wetland supply influence) can be estimated (Figure 5.12).The overall estimated linear wetland sediment detention efficiency, calculated from all release results from Network 1 and 2, illustrates that >80% is achieved for, on average, the first 4 weeks of monitoring, while average sediment detention efficiencies remain above 50% in general for the entire 52 week monitoring period. Compared to the wetland, the linear wetland's overall sediment detention rate of decrease over the 52 weeks after release is shallow, while still resulting in an ongoing decrease in detention over time/multiple rainfall-runoff events. This suggests that the linear wetland provides a more stable or consistent level of fine sediment detention with a higher detention rate (of recorded material entering the linear wetland) when compared to the wetland.

5.4.2 Sediment deposition locale

Analysis of each sample location within the linear wetland allowed identification of deposition trends within the linear wetland. Tagged sediment was found at every sample location. The total mass of tagged sediment deposited at each location (in each sample) was calculated and is presented graphically in Figure 5.13.

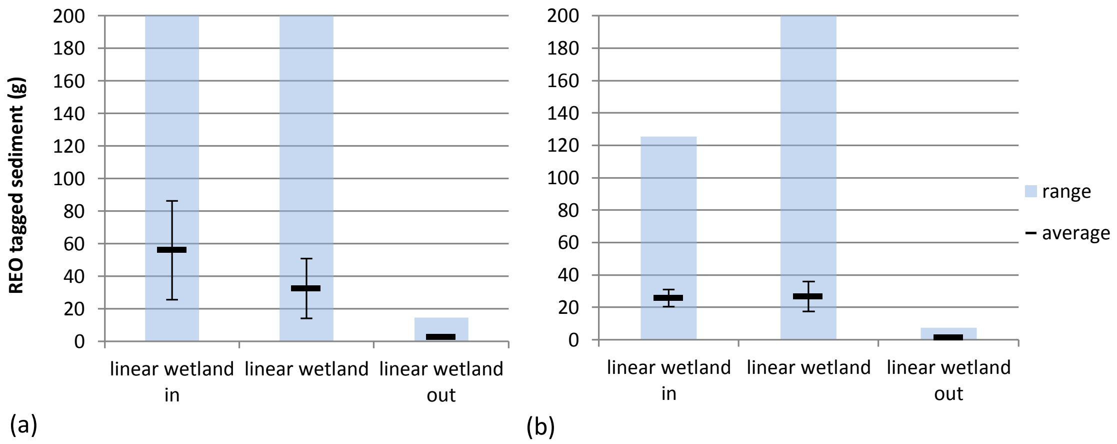


Figure 5.13 Network 1 (a) and Network 2 (b) Linear Wetland preferential deposition locations. REO tagged sediment results are representative of sample period deposition across the sample specific SuDS area. Sediment deposition range maxima for the suspended sediment (inlet and central area) reach 1147g and 705g respectively. The bed deposition range for the central cell of the linear wetland extends to 342g. The vertical axes of Figures 5.13 a and b have been limited to 200g to provide visualisation of the average and standard deviation results.

Tagged sediment was found at every sample location along the linear wetland, as illustrated in Figure 5.13, indicating its transport to, through and from the linear wetland SuDS asset. The greatest deposition mass occurs at the linear wetland inlet, decreasing through the linear wetland central section and showing the lowest average deposition at the outlet. Fine deposition at this inlet location may be due to aggregated floc particulates, allowing finer sediment material (silt and clay) to settle in the moderately turbulent inlet location. The order of magnitude of average mass (<60g) and trends (decreasing with distance to <10g at the outlet) in Networks 1 and 2 are similar, hence a reasonable conclusion to be drawn is that linear wetland design maximizes detention at the inlet with the dense and emergent (above water level) vegetation growth within the asset actively functioning to filter particles, lower flow velocity and minimize the amount of tagged sediment reaching the outlet for discharge. This supports the data of Figure 5.11 which, generally, indicates this asset to operate relatively efficiently in terms of longer term detention.

5.4.3 Shift in particle size distribution

Mastersizer S analysis of the particle size distribution of the all samples taken from the linear wetland illustrates a similar trend to that found in the deposition results (Figure 5.14).

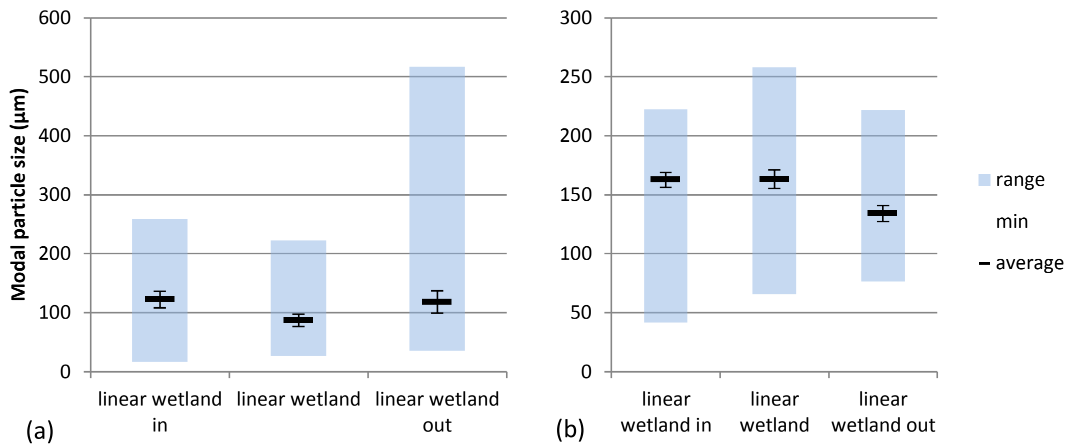


Figure 5.14 Modal particle size for linear wetland surface (a) and bed deposition (b) sediment samples.

Average suspended sediment size varies slightly within the asset across a range of 86-121 µm. This is smaller (in modal sediment size) than deposited sediments (average 135-165µm), as commensurate with the shear stress requirements to maintain particles in suspension. Only bed deposits show a distinct gradual inverse relationship between grain size and distance from inlet. The decrease in modal particle size of deposited material in the linear wetland moving downstream is potentially due to (increasing) aggregation of clay and silt (cohesive) particles along the flow oath and thus deposition of aggregated cohesive sediment further through the linear wetland This does suggest that the linear wetland functions as an effective fine sediment filter.

5.4.4 Linear wetland conclusions

A summary of the linear wetland sediment conveyance and detention efficiency research findings are provided. The key new science reflecting linear wetland fine sediment transport discussed in this section of Chapter 5 are summarised below.

Fine sediment detention is illustrated to move through the linear wetland over the monitored 52 week period.

Initial tagged sediment detention is >90%. 80% sediment detention efficiency was found to persist for (on average) 4 weeks, with the average linear wetland results achieving sediment detention efficiency >50% for the 52 week monitored period.

The linear wetland has a relatively constant average rate of decline in fine sediment detention over multiple rainfall runoff events/monitoring period.

Tagged fine sediment was found to deposit across the entire linear wetland, illustrating that tagged sediment does move through the asset over multiple rainfall-runoff events.

The amount of tagged sediment within the suspension and deposited in the linear wetland decreases down the flow path. In correlation with this, the particle size of suspended and deposited sediment also decreases down the flow path. The linear wetland is therefore found to function as a semi-porous filter, effectively removing tagged sediment from the stormwater as it moves through the linear wetland 'filter' flow path.

5.5 Swale

Networks 1 – 3 all incorporate a swale within the SuDS treatment train. Networks 1, 2 and 3 incorporate the long swale. Network 3 include two swales, a shorter swale collecting stormwater runoff from a road area which is then conveyed into a longer swale prior to discharge into the pond. Sample locations within the swales are illustrated in Figure 5.15.

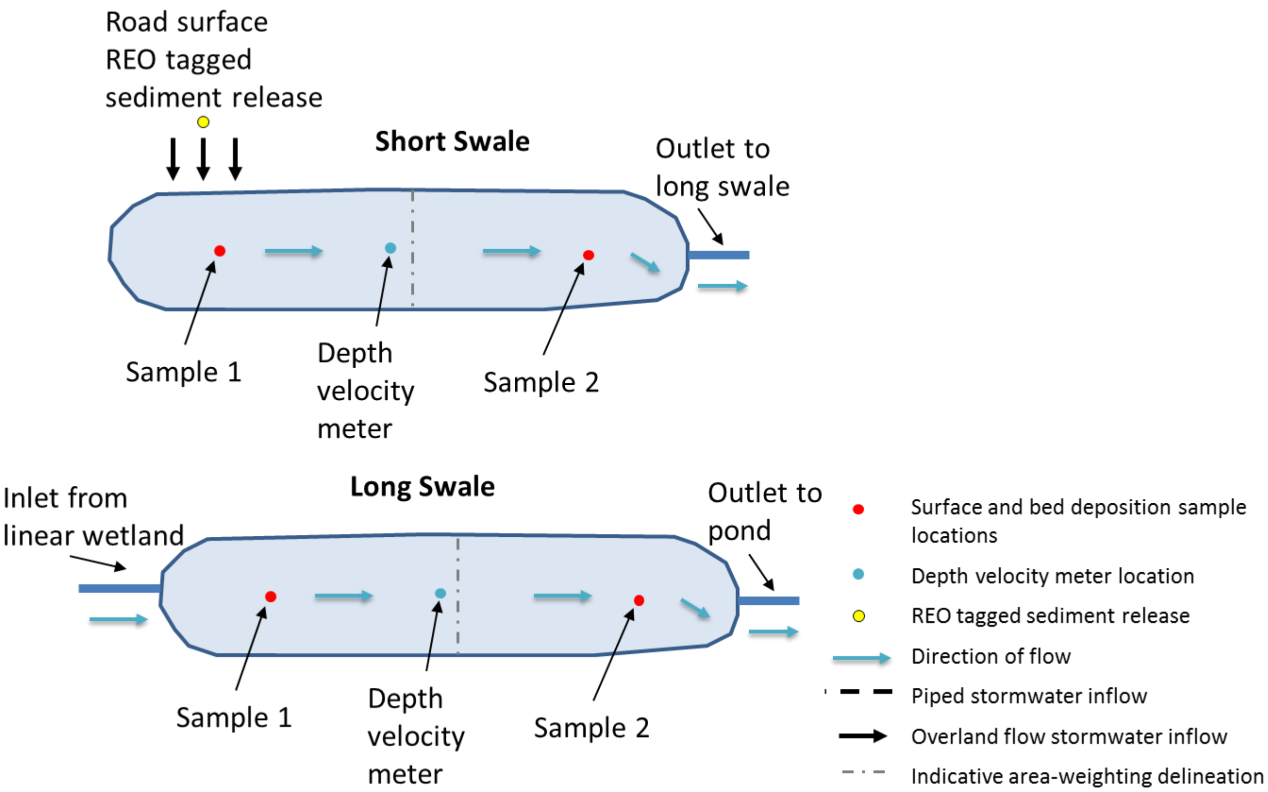


Figure 5.15 Schematic of swale sample locations

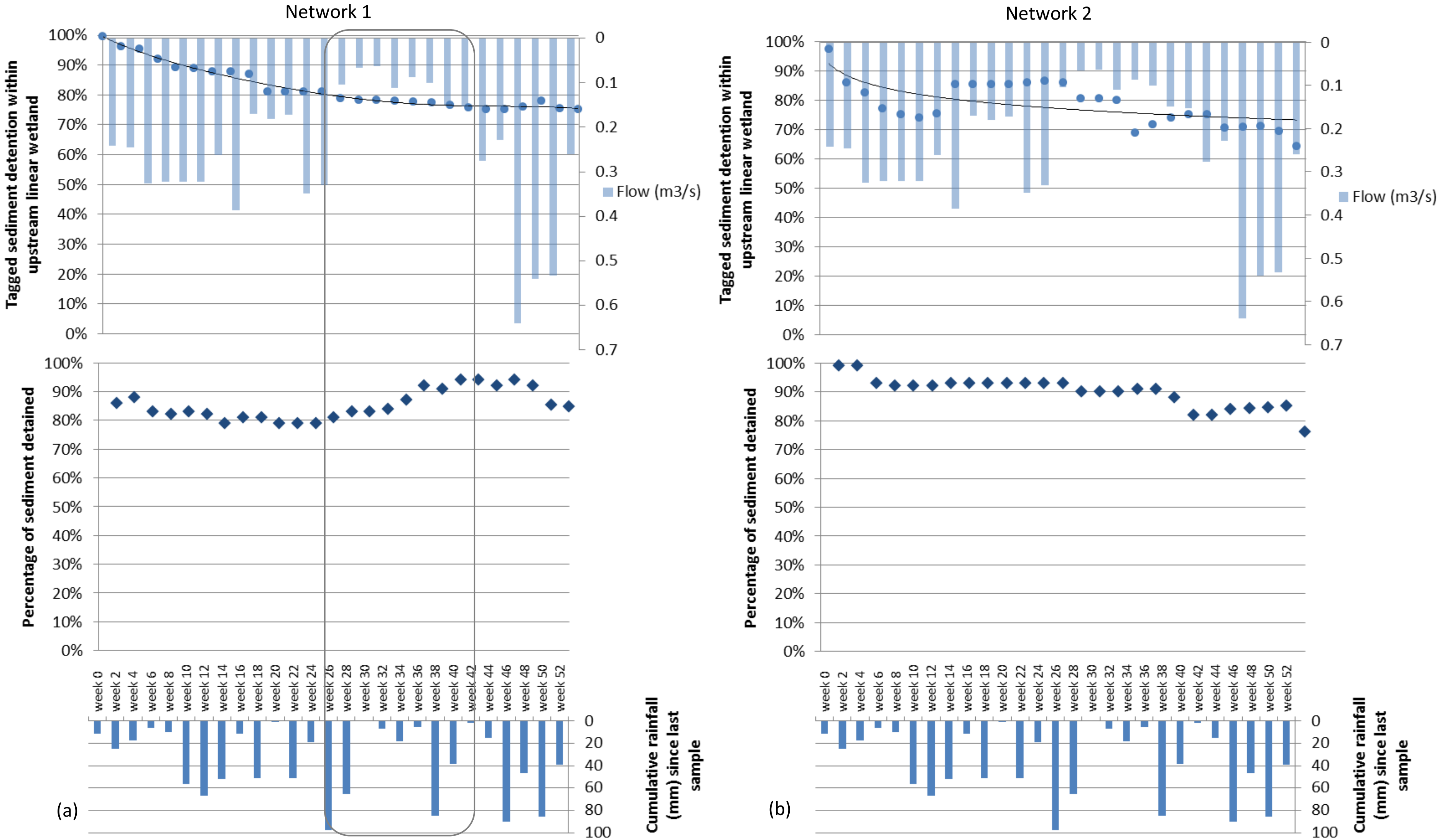
5.5.1 Sediment detention efficiency

Tagged sediment transport and deposition was monitored through the swales in

Networks 1, 2 and 3 over the 52 week monitoring period. Networks 1 and 2 include the long swale (third SuDS asset in the stormwater networks) while Network 3 is comprised of two swales (short swale conveying stormwater to the long swale) (Chapter 3, Section 3.3). The total tagged sediment temporarily detained within the swales at each monitoring period (identified in Figure 5.15, additional indicative area-weighing information is provided in Appendix V) was calculated using the basic mass balance approach. The fine sediment over the total monitoring period has been calculated and is presented for each swale in Figure 5.16. Whilst the wetland and linear wetland SuDS assets indicated a clear loss of sediment from the assets over time, the swale results suggest a relatively low percentage loss over time. It is possible that the monitored swales sediment detention efficiency remains steady or improves over time (first 20-30 weeks), due to the increased upstream conveyance/detention potential with greater monitoring duration. It is also possible that the slow but constant conveyance of tagged sediment from the upstream SuDS assets (linear wetland and wetland in Network 1 and 2) into the downstream long swale results may act as a supply limitation and thus influence the overall trend results of the long swale.

Network 1 and 2 long swales indicate a strong steady rate of detention and time after release of REO tagged Release 1 sediment. Network 1, after an initial decrease (weeks 0-12), illustrates an increased detention % from 79% to >90% over the subsequent 14 weeks (>80% in weeks 32-46). This initial relatively elevated detention may occur as a result of consistently increasing upstream tagged sediment supply. The subsequent detention increase may result from both a limited and steady upstream sediment supply (upstream detention remains constant, ~78%) and 8 weeks of low (<20mm) fortnightly rainfall (weeks 30-36, Chapter 4, Table 4.1) and thus limited flow due to the ephemeral nature of the swale. The decline in sediment detention efficiency illustrated in the final 8 weeks of monitoring occur concurrently with sizable rainfall events and thus are expected to be a rainfall-runoff response.

Network 2 illustrates a very slow decline sediment detention over the first 32 weeks of monitoring, achieving sediment detention efficiency ≥90%. The slow decline in sediment detention efficiency in Network 2 corresponds to the high but slowly declining sediment detention efficiency of the upstream (sediment source) linear wetland. The continued decline in detention may result from cumulative rainfall-runoff events, causing re-suspension and conveyance of temporarily detained sediment within the long swale.



Network 1
Tagged sediment detention within upstream linear wetland
Flow (m3/s)
Percentage of sediment detained
Cumulative rainfall (mm) since last sample
(a)
Network 2
Tagged sediment detention within upstream linear wetland
Flow (m3/s)
Percentage of sediment detained
Cumulative rainfall (mm) since last sample
(b)

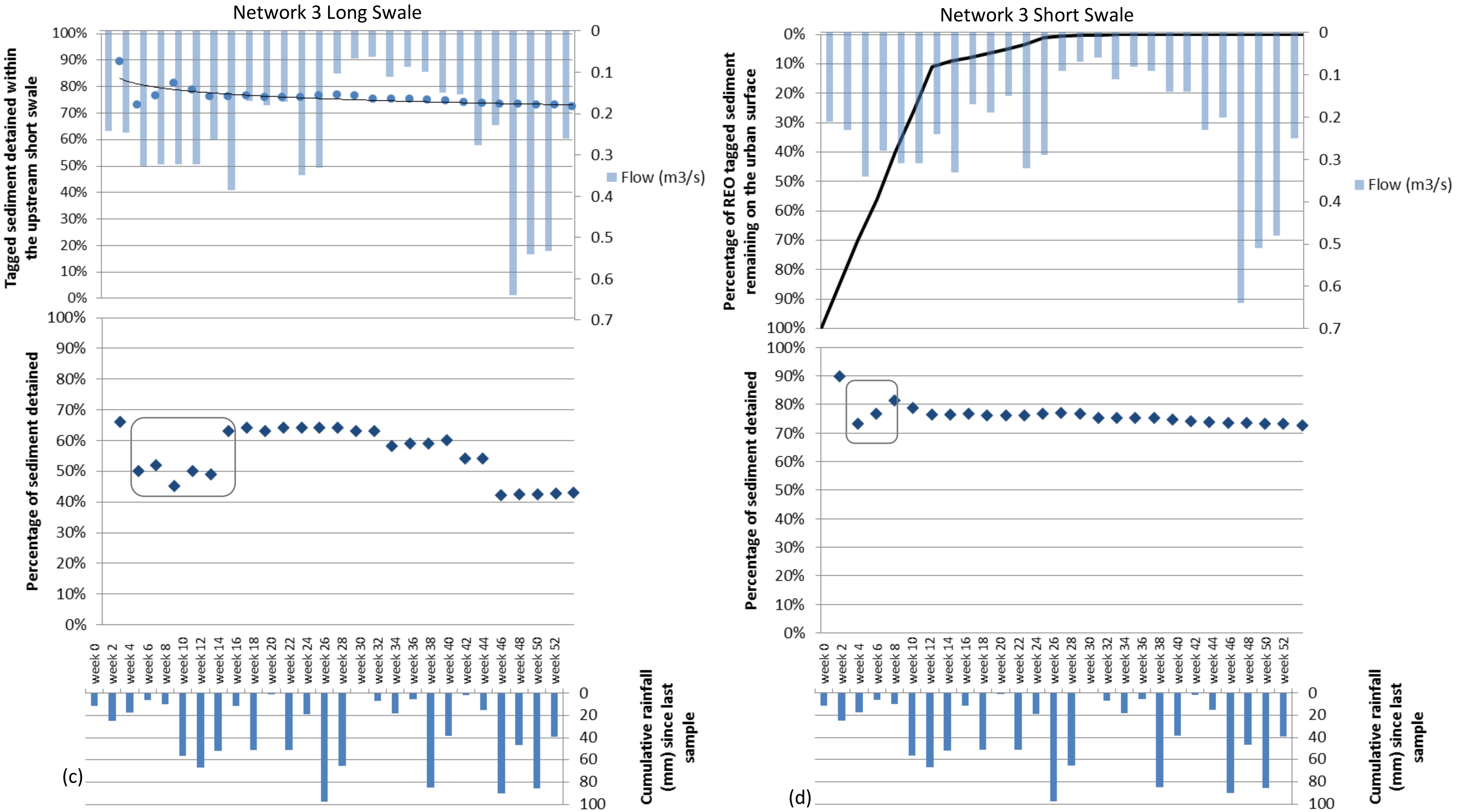


Figure 5.16 Sediment detention efficiency (%) for the long swale: release 1 for (a) Network 1, (b) Network 2, (c) Network 3; and the short swale (d) in Network 3.

The long swale trend within Network 3 shows a sharper initial and overall decline compared to Network 1 and 2. The initial detention decline to some degree mimics the supply (upstream short swale) detention efficiency while the subsequent detention declines (week 28-30, 36-38 and 40-42) occur concurrently after low to no flow period (week 30: 0mm rainfall since last sample; week 36: 5.6mm rainfall and week 42: 1.6mm rainfall). Both Network 1 and 2 long swales are downstream, and thus supplied sediment by, the wetland and linear wetland flow path sequence. This shift from the high efficiency (>80%) and shallow decline of Network 1 and 2 to the more defined decline in Network 3 long swale illustrates the influence of upstream temporary sediment detention by multiple and or wet SuDS assets (Networks 1 and 2) compared to a single/ephemeral asset (Network 3), suggesting that there is a benefit in incorporating multiple assets/wet assets upstream of a swale (greater resulting swale detention).

The short swale has no upstream SuDS assets. There is an initial increase in detention efficiency (week 2-6), potentially due to flushing of easily moved fine sediment off the urban source surface and through the swale (thus greater quantity of sediment conveyed through rather than detained within the swale) and shorter duration since release thus lower cohesive aggregation activity period. It is noted that this increase is mirrored in the long swale (receiving SuDS asset downstream from the Short Swale), illustrating a continuance of the trend downstream. Sediment detention efficiency in the short swale is found to decrease slowly over the monitoring period (after week 6) while the tagged sediment released from the tagged urban surface (the road) declines steeply (over the first 12 weeks). The correlation between the short swale sediment detention efficiency and the upstream road tagged sediment release is positive ($r = 0.37$). This illustrates that as the quantity of sediment remaining on the road surface decreases (and thus a larger total quantity of tagged sediment enters the short swale) the sediment detention efficiency of the short swale declines. This suggests that the short swale may also be influenced by supply, in conjunction with rainfall and flow events.

Repetition of the tagged sediment experiments (three monthly uniquely tagged sediment releases) provided a more comprehensive dataset from which to analyse fine sediment detention trends through the swales in Networks 1, 2 and 3. Figures 5.17(a-d) illustrate the tagged sediment detention for all 4 experiment repetitions within each of the swales.

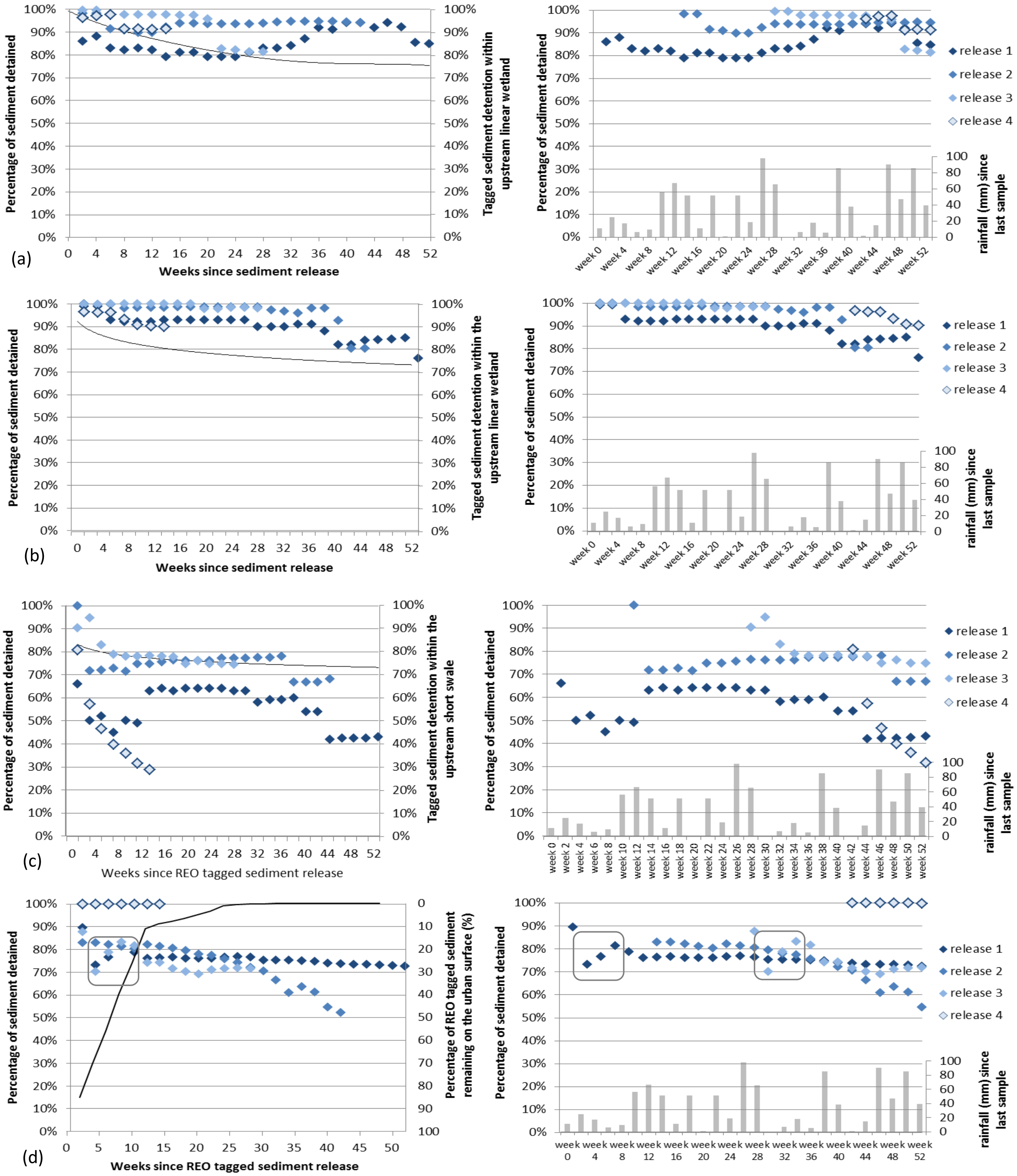


Figure 5.17 Tagged fine sediment detention through h the long swale for Network 1 (a) and Network 2 (b), Network 3(c) and the short swale on Network 3 (d). The left graphs in each of these figures presents REO tagged sediment detention from the time of sediment release, while the right graphs illustrate the detention % relative to the date of sampling. Figure 5.16 (d) Release 4 results range between 98-99%.

Repeated experiment results for the long swale in Networks 1 and 2 show comparable trends across the all the releases (Figure 5.17 a and b). Both of these long swale results datasets illustrate a high sediment detention efficiency over the majority (first 36 weeks) of the monitoring period. A decline in sediment detention efficiency is found, consistently across all four releases on both Network datasets (Networks 1 and 2) over the final few weeks of monitoring. This decline occurs during high a high rainfall-runoff period (all the last 8 weeks of monitoring have above average rainfall, with weeks 46 and 50 illustrating fortnightly cumulative rainfall depth > 80mm). The late decline in detention in both Network 1 and 2 are thus suggested to result from rainfall influences. The results from Network 1 and 2 provide a strong consistent trend, suggesting with some confidence that the long swale provide initial effective detention efficiency but that there is a decline in detention % after an extended monitoring period.

Network 3 long swale presents a disparate general trend from Network 1 and 2. There is a steeper decline in sediment detention across all releases, with the detention generally falling below 80% after week 4 (Figure 5.16 c). The greater range and decline in detention for Network 3 long swale suggests a definite shift in long swale functionality compared to that illustrated for Networks 1 and 2. The lower overall detention efficiency and greater variance between release results suggests a less constrained upstream sediment supply and/or greater sensitivity of this tagged sediment mass to rainfall-runoff results, supporting the suggestion of multiple/wet upstream SuDS asset benefit to swale detention efficiency.

The short swale, within Network 3, illustrates a consistent decline in sediment detention from the initial tagged sediment release onto the urban (road) surface until cessation of monitoring (with the exception of Release 4). Release 1 and 3 illustrate an initial decline-rise-decline trend (highlighted in Figure 5.17 d), potentially resultant from an increase in rainfall-runoff (from no/low flow conditions - causing high initial detention, to moderate flow conditions- resulting in re-suspension and conveyance or potentially from a lower number of events occurring) over the initial influx of (tagged) sediment into the short swale (i.e. when sediment influx first starts, prior to longer term potential burial). Release 4 presents anomalous results, with very high detention % for the full (12 week) monitoring period (98-99%). The consistently elevated detention results, not found for this extent within any other monitored SuDS asset, may be due to a limited supply (i.e. of the majority of Release 4 become detained in the overland flow path from the road to the short swale thus not entering the swale or a change in urban surface management such a road sweeping that might remove tagged sediment thus reducing the overall load into the short swale). The exceptional detention efficiency may also be due to modified traffic use, resulting in conveyance of tagged sediment away from the swale (on tyres or car extrusions), compaction and therefore limited release or addition of grit/spillage and associated clean-up/change in surface management potentially causing a blinding or burial or Release 4 within the road release area.

The average trend, variance and standard error (as an analysis of trend representation accuracy) for each monitored Network swale is presented in Table 5.3

Table 5.3 SuDS asset sediment detention efficiency (%)

Monitoring period	release	week 2	week 8	week 16	week 24	week 32	week 40	week 48	week 52
Long swale - Network 1									
average	0	96%	90%	91%	85%	91%	94%	85%	84%
variance		4%	5%	7%	6%	4%	5%	<1%	0%
StError		1%	1%	2%	1%	1%	1%	<1%	0%
Long swale - Network 2									
average	0	99%	95%	97%	97%	93%	81%	85%	79%
variance		2%	4%	3%	3%	2%	1%	0%	0%
StError		0%	1%	1%	1%	1%	<1%	0%	0%
Long swale - Network 3									
average	0	68%	59%	72%	72%	38%	60%	43%	45%
variance		17%	17%	7%	6%	9%	6%	0%	0%
StError		4%	4%	2%	1%	2%	2%	0%	0%
Long swale – average									
average	0	88%	82%	87%	85%	84%	79%	71%	69%
variance		17%	19%	12%	11%	13%	14%	20%	18%
StError		6%	7%	4%	4%	5%	5%	7%	7%
Short swale - Network 3									
average	0	82%	85%	75%	74%	68%	63%	73%	71%
variance		12%	9%	4%	2%	7%	11%	0%	0%
StError		3%	2%	1%	1%	2%	3%	0%	0%
Swale - average									
average	0	86%	82%	84%	82%	80%	75%	72%	69%
variance		16%	17%	12%	11%	14%	15%	17%	16%
StError		6%	6%	4%	4%	5%	5%	6%	6%

Review of all tagged sediment transport progress through the long and short swales over the 12 month monitoring period provides further insight into the generalised multiple-event swale sediment detention efficiency. The detention efficiencies for each swale within Network 1-3 are presented in Table 5.3, will all swales trends illustrating a decline in sediment detention efficiency over the monitoring period. The standard error of the mean for all calculations is small, <10% and the variance in Network 1 and 2 are small. The greater variances are seen in Network 3, potentially due to greater sensitivity to rainfall-runoff and sediment supply as a result of the shorter SuDS flow path (number of SuDS assets upstream of the swale). In general, the low standard error results illustrate confidence in the asset Networks specific and average swale trends presented in Table 5.3.

Using the total long and short swale datasets, with multiple experimental repetitions, an indicative overall sediment detention trend can be calculated using dataset averages. Figure 5.18 shows the range of sediment detention values within the entire dataset and the indicative average trends for both the long and short swale over the monitoring period.

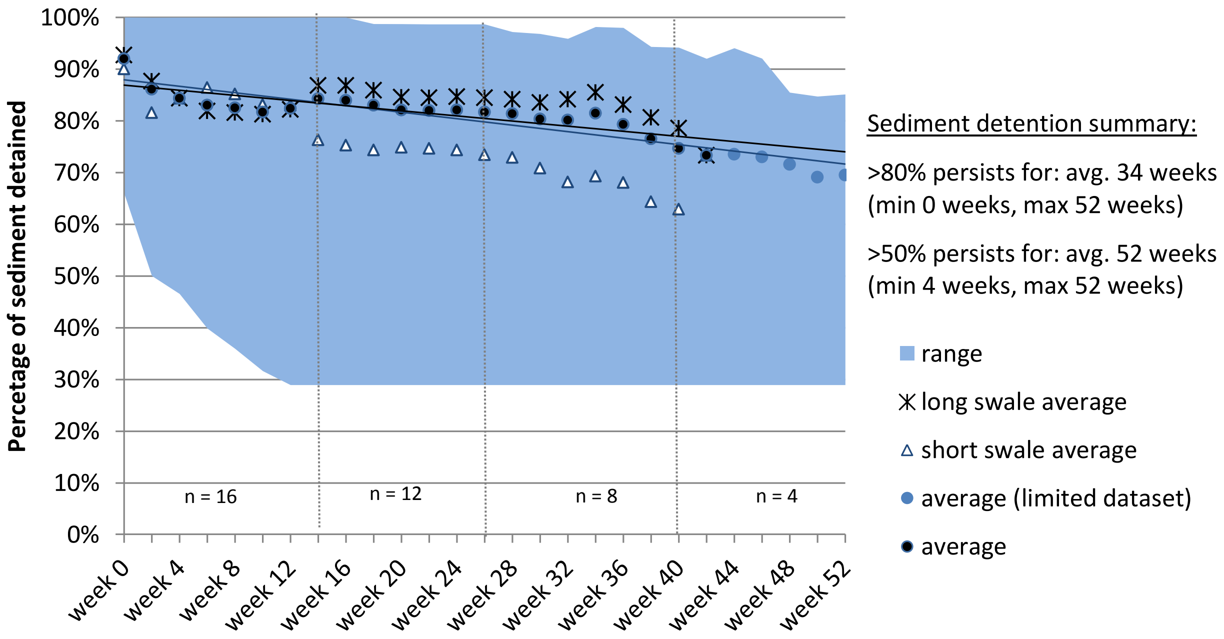


Figure 5.18 Average swale sediment detention trend. The black points and trend line indicate the average wetland sediment detention efficiency trend determined from multiple release results (Release 1-4). The blue points and trend line illustrate the final 10 weeks of Release 1 (the longest monitored REO tagged sediment release) and do not have the equivalent sampling replication as the black data points (n=4 for blue points, n=16 for black points).

When all the experimental repetitions are considered a shallow but consistently decreasing trend can be extrapolated from the field monitoring results. Figure 5.18 illustrates the trends for both the long and short swale, illustrating the higher sediment detention efficiency for the long swale (potentially due to upstream SuDS flow and sediment management). Both illustrate moderately efficient sediment detention efficiencies, above 50%, with the short swale achieving high efficiency (>80%) for the first 14 weeks while the long swale illustrates 38 weeks of high detention %. When combined, to provide a general swale detention efficiency analysis, the trend suggests that high (>80%) efficiency may be achieved for 34 weeks after sediment release, with greater than 50% detention achievable for 52 weeks. All trends presented in Figure 5.18 illustrate a consistent shallow decline in sediment detention efficiency over the total monitoring period, the influence of multiple rainfall-runoff events and resultant sediment conveyance through the monitored swales.

5.5.2 *Sediment deposition locale*

The swale monitoring was limited to samples at the inlet and outlet. The sediment deposited in the upper (inlet) and lower (outlet) section of the swales have been quantified and compared to identify where sediment preferentially deposits. The results are presented in Figure 5.19.

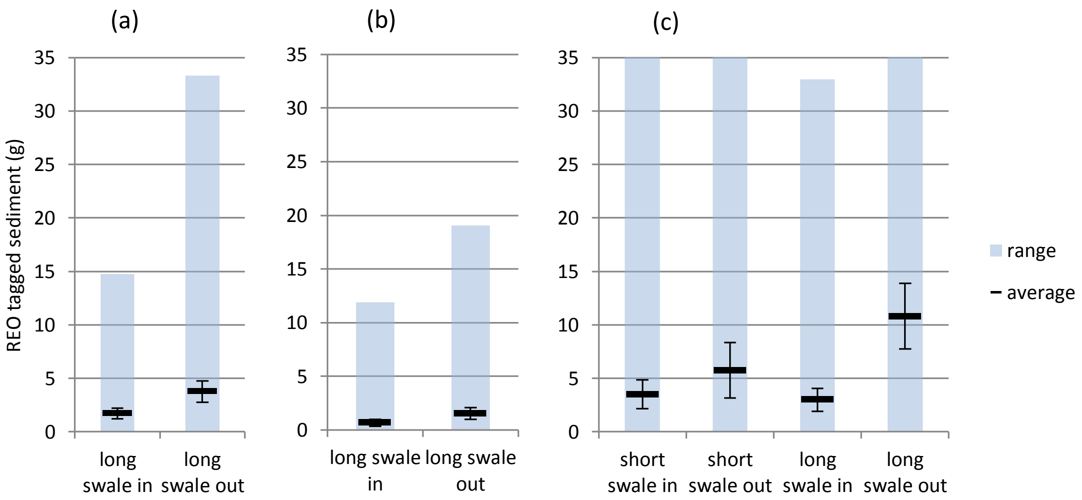


Figure 5.19 Network 1 (a), Network 2 (b) and Network (c) Swale preferential deposition locations (release 1). REO tagged sediment results are representative of sample period deposition across the sample specific SuDS area. The sediment particle size range for the short swale inlet, outlet and long swale outlet within Network 3 extends to 47g, 95g and 105g, but Figure 5.19 (c) vertical axis is limited to 35g to provide average and standard deviation result visualisation.

For all swales, the amount of tagged sediment deposited at the downstream extent of the swale was greater than the amount deposited at the upstream extent. Thus the swale detention efficiency is suggested to increase moving downstream through the swale. The long swale shows a 150-250% increase in sediment deposition at the downstream extent of the swale, short swale downstream deposition increases 65%. The difference in downstream detention is suggested to occur as a result of swale design and cohesive properties of fine urban sediment. The long swale design has less dense and shorter vegetation (mown grass) in comparison to the short swale (grass and shrub mix) providing lower flow resistance and thus lower overall detention capacity; while being twice the length (long swale - 330m is twice the length of the short swale – 151m) thus providing a longer flow path over which to detain and allow aggregation of fine sediment. The short swale, in Network 3, discharges directly to the long swale, while the long swale discharges to the J4M8 pond resulting in a standing water level at the downstream extent of the long swale. The downstream boundary conditions (dry for the short swale, wet for the long swale) may result in greater proportional deposition in the long swale (in comparison to upstream long swale deposition) due to the flow velocity decrease and thus greater settling potential occurring when long swale discharge meets the wet downstream boundary. Furthermore, the swales have channelized inflow resulting in higher shear stresses at the upstream extent. The vegetation resistance within the swales acts to reduce flow velocity and carrying capacity leading to greater deposition at the downstream swale outlet. The sediment deposition occurring in Network 3 is expected to result from the shorter total stormwater flow path (road surface – short swale – long swale) compared to Networks 1 and 2 (which incorporate a wetland and linear wetland upstream from the long swale).

5.5.3 *Shift in particle size distribution*

The particle size of surface and bed deposition sediment samples was analysed. Samples were monodisperse in nature, and the primary particle size for all samples was identified to allow total swale sample location particle size analysis. The average overall primary particle size for the upstream and downstream sediment samples in both the short and long swales are presented in Figure 5.19.

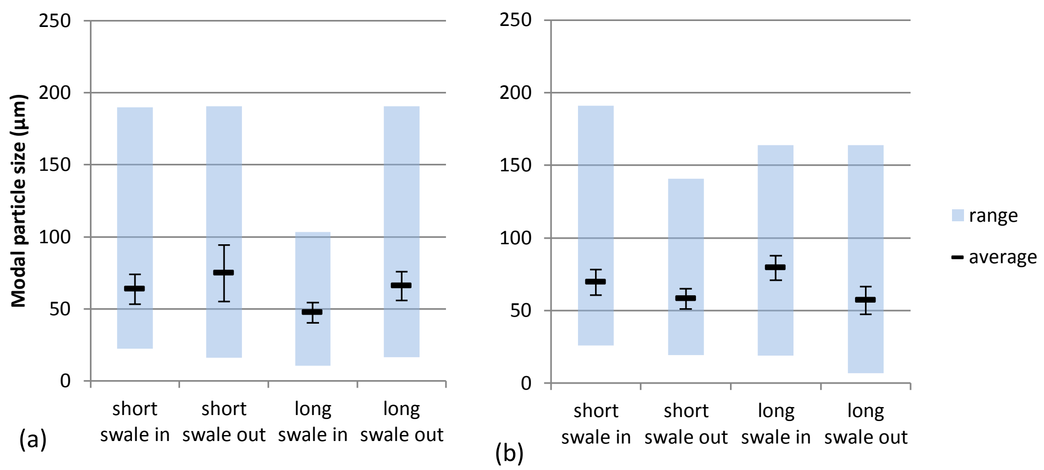


Figure 5.20 Modal particle size for the swale surface (a) and bed deposition (b) sediment samples.

Modal particles size for all long swale samples were analysed together, providing the long swale modal particle size average, standard deviation and range for suspended (Figure 5.20a) and deposited (Figure 5.20b) sediment. The short swale samples were analysed in the same way. Suspended sediment modal particle size is seen to increase downstream in both the long and short swales. This suggests that the swale functions to detain the finer particles, removing them from suspension through vegetation resistance or filtering (potentially in conjunction with cohesive aggregation), thus shifting the modal particle size higher downstream. Both the short and long swale illustrate this trend; a shift from 63-74μm in the short swale, 47-65 μm in the long swale. The decrease in the swale deposited modal size may be due to upstream SuDS sediment detention resulting in less larger sediment material being conveyed into the long swale and thus an overall lower modal particle size. It may also result from cohesive particle aggregation and settling, resulting in greater fine sediment deposition down the swale flow path.

5.5.4 Swale conclusions

A summary of the swale sediment conveyance and detention efficiency research findings is provided. The key new science reflecting wetland fine sediment transport discussed in this section of Chapter 5 is summarised below.

Fine sediment moves through the monitored swales over multiple rainfall-runoff events. In the short swale, sediment is less likely to be permanently detained once the supply event (from the road surface) ceases.

The swale in Networks 1 and 2 illustrate greater sediment detention efficiency than for either swale in Network 3. This may illustrate the influence of wetland and linear wetland upstream flow/sediment detention influence.

The long swale shows a shallow decline in sediment detention efficiency, with detention % >80% over the first 32weeks (average) and >50% for the full monitoring period (52 weeks).

The short swale shows a similar decline in detention %, achieving >80% detention efficiency for the first 8 weeks and >50% for 52 weeks.

The initial temporary detention efficiency of the swale, during week 0-2, was found to be high – greater than 80% of the supply for most Releases. The swale was found to achieve >80% sediment detention efficiency for 34 weeks (average).

Network 1 and 2 long swales illustrate low variance, while all swales illustrate low standard error >10% providing confidence in the trends illustrated.

Sediment detention occurs throughout the swales, with notable bed deposition near the outlet (where there is potentially lower shear stress and flow velocity).

Suspended sediment modal particle size increases through the swale, resulting in the removal of fine sediment from suspension. Bed deposition modal particle size decreases through the swale, illustrating the swales to act as effective sediment filters, with preferential detention of larger (higher settling velocity) sediment particles.

5.6 Pond

The pond results presented in this section focus on data collected from the Newcastle Great Park field site (Network 4). While J4M8 (Network 1, 2 and 3) has a pond, it was not designed as a water quality SuDS asset. Furthermore, the size and depth of this asset is too large to safely allow detailed monitoring for sediment transport and the outlet of the J4M8 pond is a small sluice gate (a metal, mechanically opened/closed culvert). Thus, the inlet and outlet of the pond in Networks 1, 2 and 3 have been monitored for inclusion in the mass balance analysis for these networks and are presented for comparison in this chapter. However, detailed internal pond sediment transport analysis has been undertaken using the NGP field site (Network 4) as internal pond samples of bed deposition and surface flow were collected throughout the pond. Schematics of the J4M8 and NGP pond sample locations are presented in Figure 5.21 (additional indicative area-weighing information is provided in Appendix V).

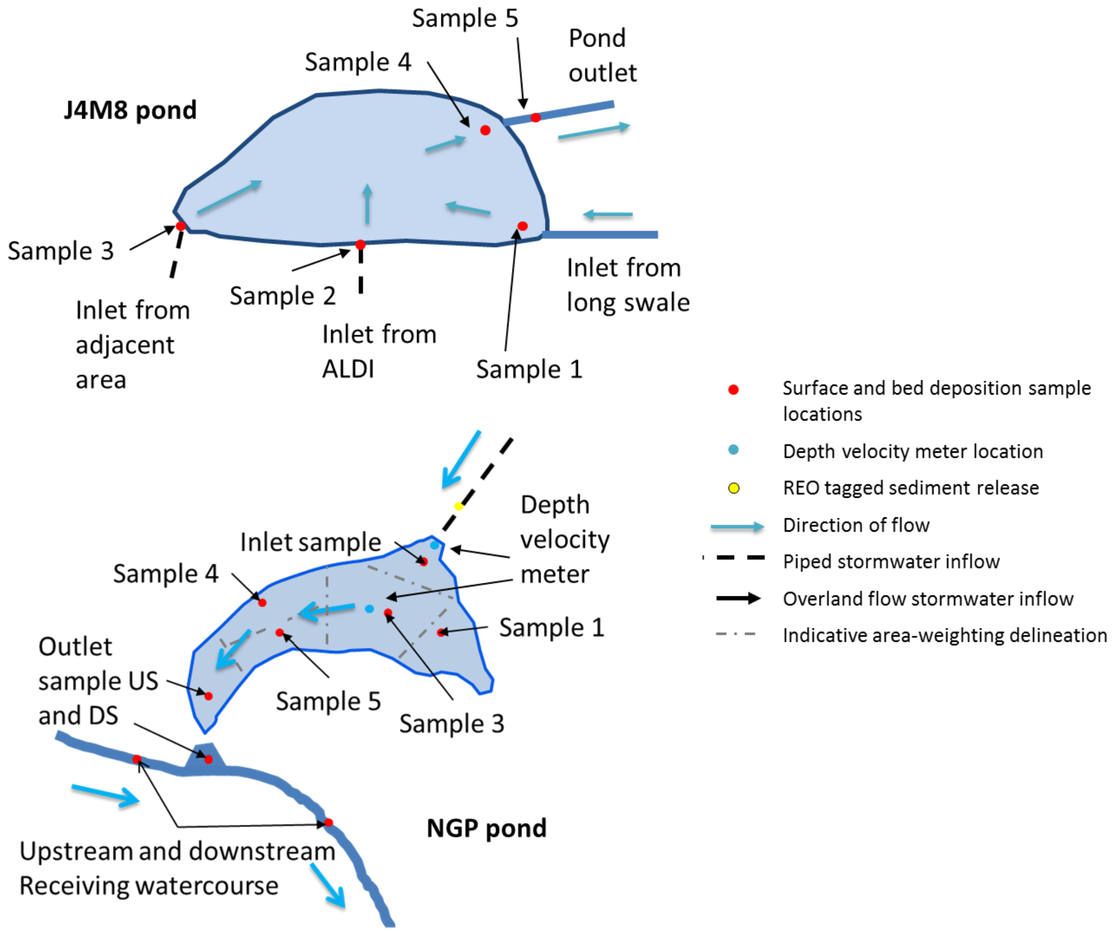


Figure 5.21 Schematic of pond sample locations

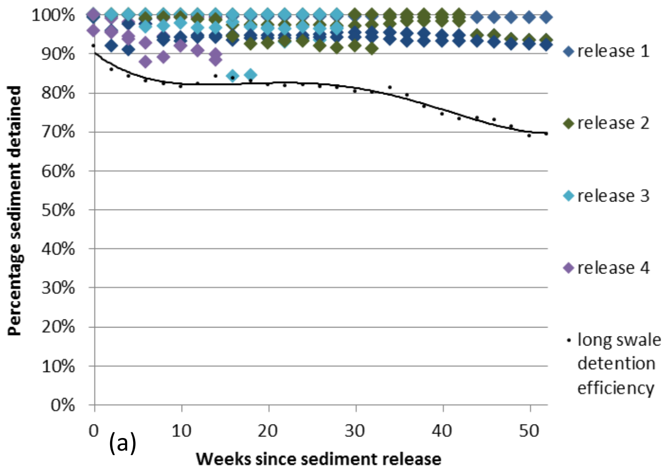
5.6.1 *Sediment detention efficiency*

Within the J4M8 pond (16240m^2) sediment was collect at all four inlets; the inlet from the long swale (receiving tagged sediment from Networks 1, 2 and 3) and the two adjacent sub-catchment areas. Sediment tracing through the NGP pond (2400m^2) occurred on cessation of Network 1, 2 and 3 monitoring. Thus the tagged sediment experimentation across the NGP pond was able to incorporate advancements, individual particle size REO tag and tracing, in sediment tracing and analysis learnt from the field work previously undertaken.

J4M8 Pond

A simple analysis of inflow versus outflow from the J4M8 pond is presented in Figure 5.22. The outlet results, sampled downstream from the sluice outlet of the pond provided an insight into the detention occurring within the overall pond. Due to the extensive size and depth of this pond, in conjunction with the outlet control (outlet void

~ =100mm) very limited tagged sediment was expected to occur in the downstream suspended bed sediment trap samples.



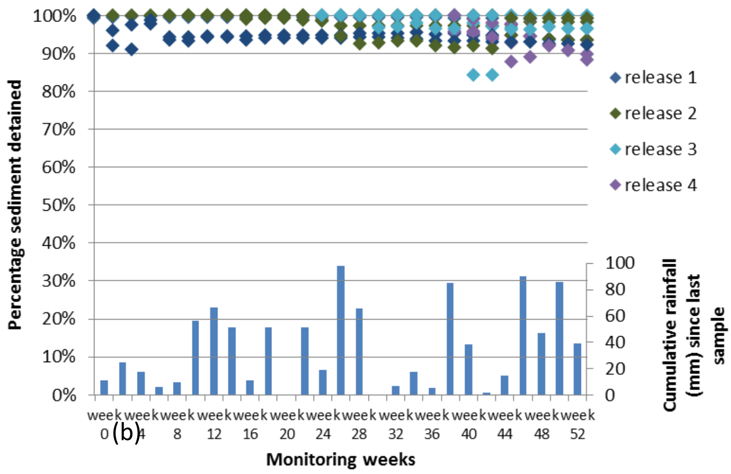


Figure 5.22 Tagged sediment detention efficiency within the J4M8 pond. Release 1, 2, 3 and 4 are presented in blue, green, turquoise and purple, with the upstream long swale sediment detention efficiency illustrated in black. Figure 5.22(a) presents the J4M8 pond sediment detention efficiency after initial tagged sediment release, while Figure 5.22(b) presents the detention % at the date of sampling (time stamped) and the corresponding rainfall.

The sediment detention results presented in Figure 5.22 illustrate that the J4M8 pond, while not designed as a SuDS asset, provides effective detention of any sediment released from the upstream networks (Networks 1, 2 and 3). The detention efficiency for all releases (Release1-4) was >85%, with the majority of results >90%. The consistently high sediment detention efficiency is attributed to the significant size of the pond and the constrained outlet orifice (causing any sediment that might reach the pond outlet to settle behind the outlet). However, despite the consistently high detention, some tagged sediment (>15%) is passing through this large pond. This suggests that a small proportion fine sediment (potentially the very fine fraction that remains in suspension for extended periods) entering even a very large pond may continue to move.

NGP Pond

One of the key questions arising from analysis of Network 1, 2 and 3 sediment monitoring was the capacity for fine sand (~150µm) versus clay (2-5 µm) sized particles transport through SuDS. As illustrated in Figures 5.8, 5.14 and 5.20, each of the J4M8 monitored SuDS assets appears to modify the modal particle size of both suspended and deposited sediment particle size distribution. The wetland and linear wetland (wet assets) appear to decrease the suspended sediment modal particle size (preferential detention of larger particles), while swale monitoring illustrated an increase (suggesting the detention of fine material in the flow). In general, the inverse relationship was found for bed deposition particle size, suggesting wet assets such as the wetland and linear wetlands detain courser sediment closer to the asset inlet while swale provide greater fine sediment deposition further downstream. To examine the influence of particle size on sediment detention function further clay and fine sand particles were tagged separately and monitored within the NGP pond (methodology details provide in Chapter 3, Section 3.5.9).

Tagged urban sediment, artificially released into the inlet of the pond, was monitored over a six month period. Figure 5.23 presents the sediment detention efficiency results for the total released sediment (a) and then the fine sand (b) and clay (c) components of the total artificial urban sediment release.

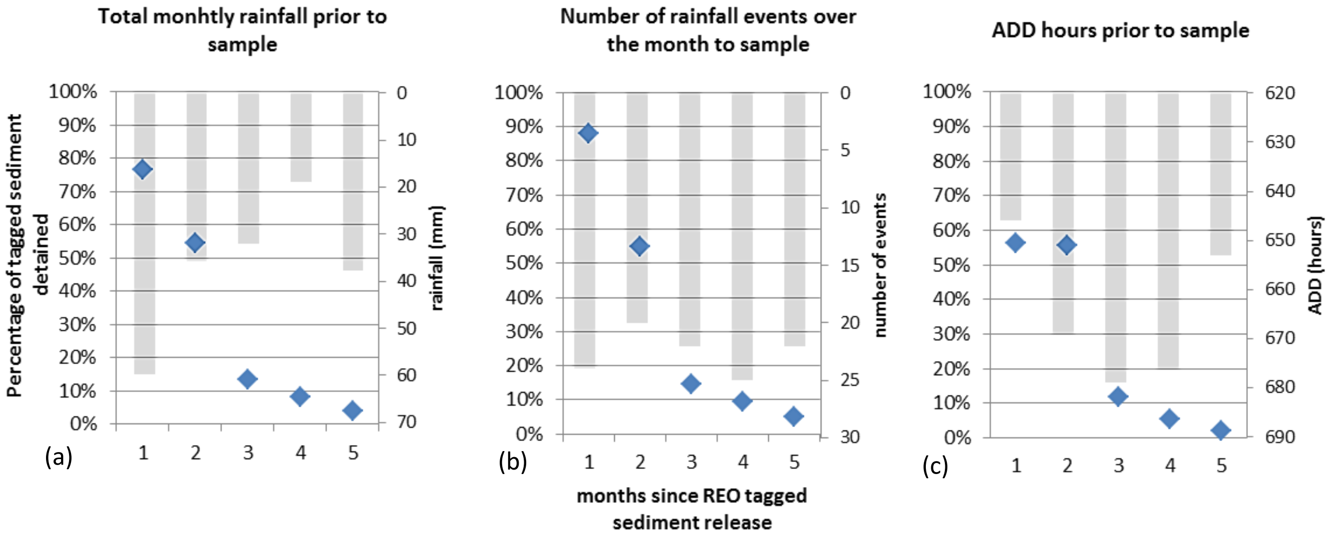


Figure 5.23 Pond sediment detention efficiency (%) for release 1 for (a) total overall sediment released, (b) sand and (c) kaolin clay.

The tagged urban sediment detained in this pond was found to decrease dramatically over the first 3 months of the 5 month monitoring period. The initial overall sediment detention was moderate, 77%, with over 80% of the fine sand detained within the pond. The clay material moved through the pond faster than sand during the first month of monitoring, with only a 56% clay detention occurring during the first month after sediment release. The initial high conveyance of clay material is expected to occur due to high suspension capacities and low settling velocities of clay particles (in comparison to fine sand). This would explain the initial 'flushing' of clay materials through the pond in the first month, while allowing remaining clay material to achieve a higher detention through vegetation resistance/trapping and settling (potentially as cohesive particulates) over the following months (similar to that of the fine sand).

Both clay and sand deposition fell below 10% after 4 months of rainfall-runoff events. Part of the reason for high transport results seen in this pond may result from the high supply caused by up upstream construction activities. The construction works (residential development) occurring in the NGP resulted in a high fine sediment influx into this pond. Thus tagged sediment results for the NGP pond may be buried or lost due to high sediment influx and deposition volumes. However, the dry weather flow into and through this pond was also found to be greater than for the monitored J4M8 SuDS assets (average inflow of $0.98m^3/s$ compared to $0.24\text{-}0.47m^3/s$ in J4M8, Chapter 4, Table 4.3) suggesting that there may be greater flow conveyance through this pond and thus potentially higher sediment conveyance. Four repetitions of the tagged sediment release experiment were completed for the pond. Figure 5.24 presents the results for the total, sand and clay sediment detention efficiencies.

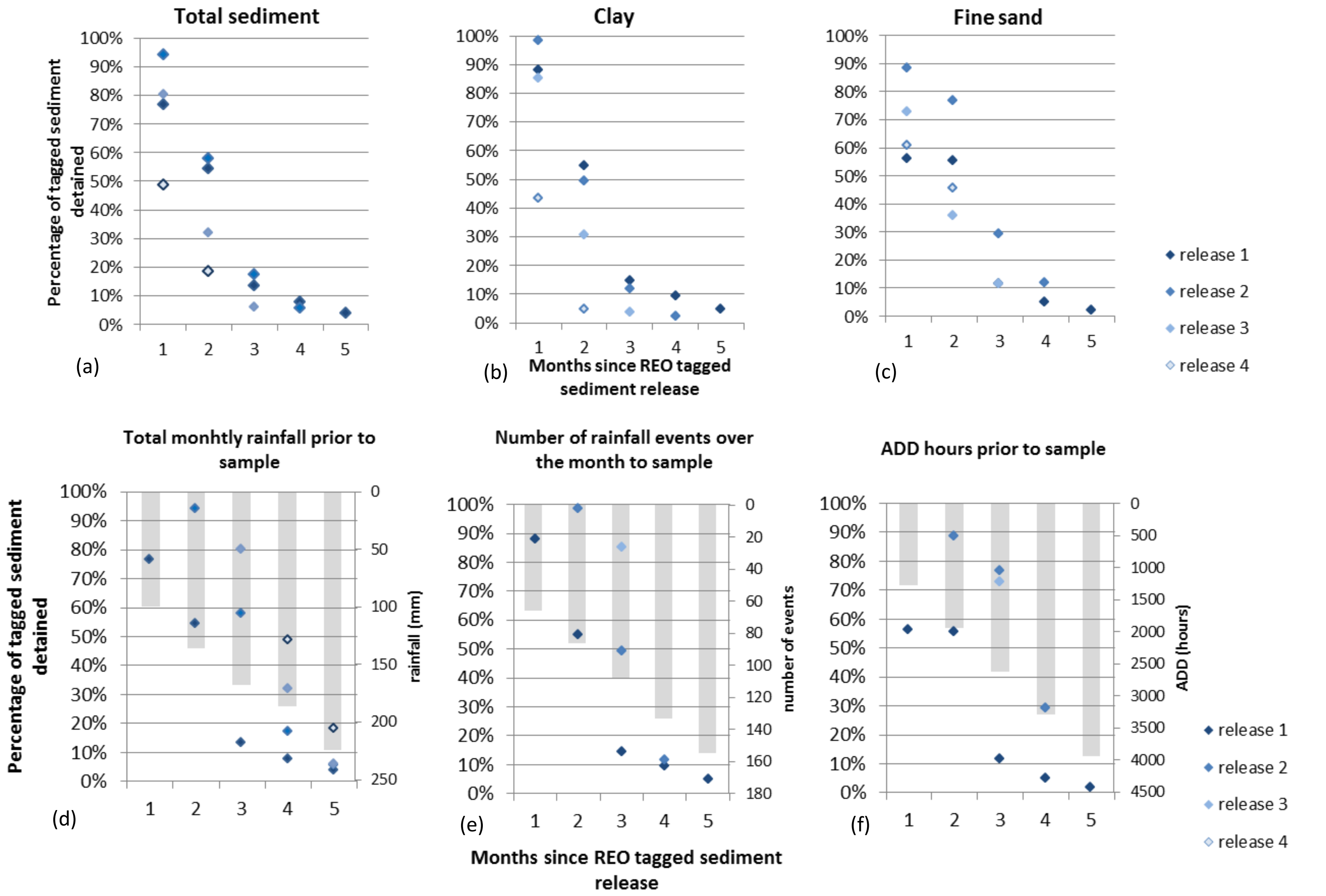


Figure 5.24 Tagged fine sediment detention through the pond, total sediment (a, d), sand (b, e) and kaolin clay (c, f). Figures 5.23 a, b and c present detention% from the time of release while Figures 5.23 d, e and f present results relative to the date of sampling, in conjunction with Cumulative rainfall characteristics.

A strong decreasing sediment detention efficiency trend is seen in all of the releases, for both clay and sand material. After 3 months of rainfall-runoff events, the sediment detention falls below 30% for all releases and sediment types (Figure 5.24 a-c). Thus the longer term, multiple event detention efficiency of the NGP pond is shown to be ineffective (<20%). It is suggested that the decline in sediment detention efficiency occurs due to cumulative rainfall-runoff events, re-suspending and conveying sediment within the pond. Figure 5.24 (e) presents the monthly rainfall event occurrence, illustrating that month 1 has the greatest number of rainfall event occurrences (42), with the following 4 months receiving 20-25 events per month. The initial decline in Release 1 may result from this high rainfall event occurrence and total rainfall (depth, mm, Figure 5.24 d). Release 2, 3 and 4 illustrate a decline in detention % with increasing cumulative rainfall events, total rainfall and dry periods (ADD) (Figure 5.24 d-f).

The sediment detention efficiency results show a repetitive steeply declining trend in detention, across all four releases and both sediment sizes (sand and clay material). The variance between release results has been calculated and presented in Table 5.4. The results illustrate the NGP and J4M8 specific pond sediment detention function, and provide an indicative pond detention efficiency result for multiple rainfall-runoff events.

Table 5.4 SuDS asset sediment detention efficiency (%)

Monitoring period	release	month 1	month 2	month 3	month 4	month 5	month 12
NGP Pond – clay							
average	0	70%	54%	18%	9%	2%	-
variance		2%	2%	1%	<1%	<1%	-
StdError		1%	1%	<1%	<1%	<1%	-
NGP Pond - sand							
average	0	79%	35%	10%	6%	5%	-
variance		4%	4%	<1%	<1%	<1%	-
StdError		2%	2%	<1%	<1%	<1%	-
NGP Pond – total tagged sediment release							
average	0	75%	41%	12%	7%	4%	-
variance		3%	3%	<1%	<1%	<1%	-
StdError		1%	1%	<1%	<1%	<1%	-
J4M8 Pond – average							
average	0	98%	97%	97%	96%	96%	95%
variance		3%	3%	3%	5%	3%	3%
Std.Error		1%	1%	1%	1%	1%	1%

The tagged sediment detention results for the pond were averaged, with the variance and standard error of the mean calculated for each sediment type (sand, clay) and the total tagged sediment release presented in Table 5.4. The initial (first month) average detention efficiency for the NGP pond was between 70-80%, declining to <10% after 4 months (16 weeks). The pond detention results show a low variance and standard error for all calculated averages (>10%), indicating that trends presented within Table 5.4 for both ponds (individually) illustrate internal data validity. Figure 5.25 visually illustrates the average trend and range of detention % for the NGP and J4M8 ponds.

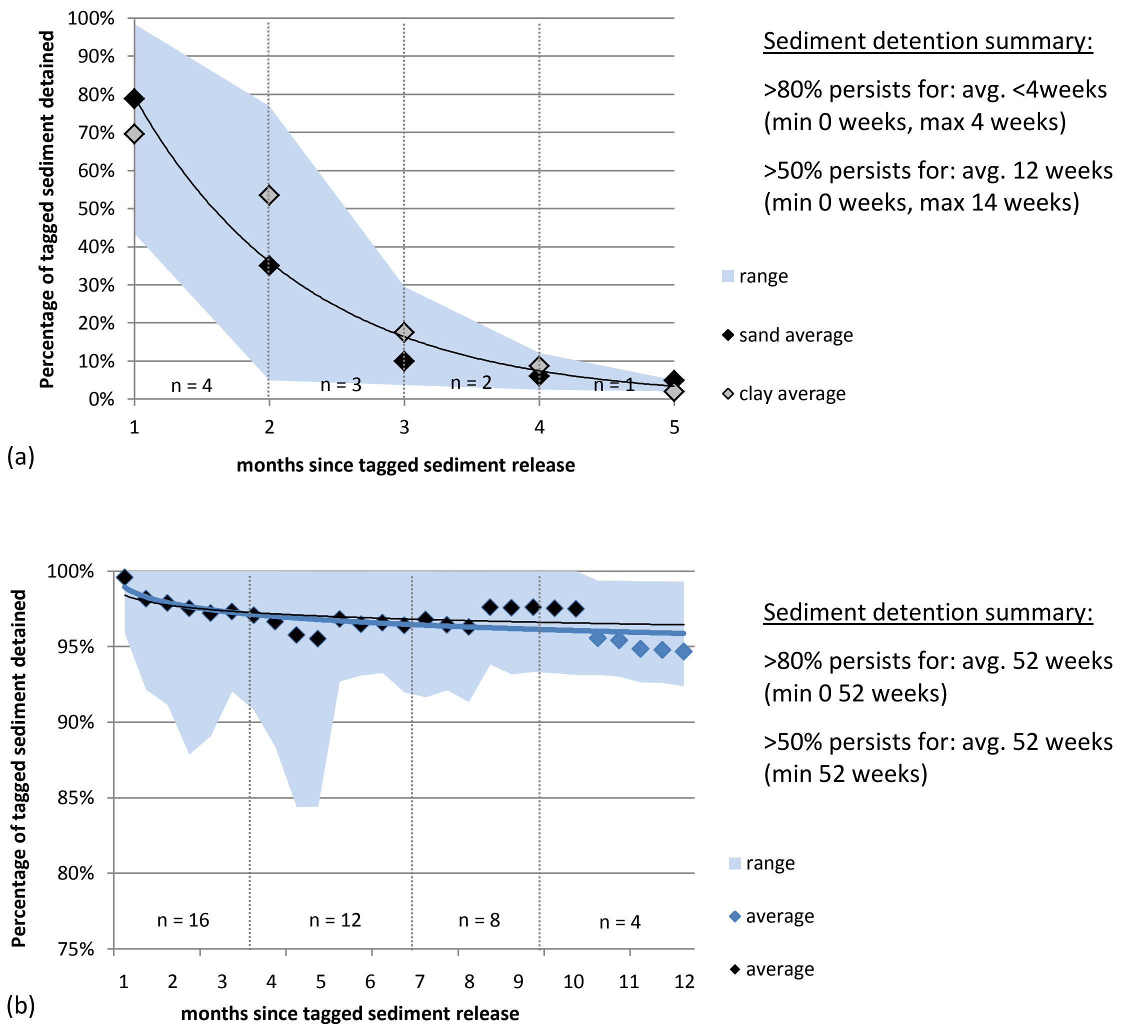


Figure 5.25 Average pond sediment detention trend for the NGP pond (a) and J4M8 pond (b). The range of results is illustrated in blue shading, while the total average trend is illustrated through the trend lines.

The J4M8 pond results illustrate up to 20% detention range around the average trend (± 20%). The average trend illustrates a high detention efficiency (>90%), with a very slow decline in detention over the 12 month monitoring period (3%). This pond shows a high detention efficiency but despite its size and outlet design a very slow decline is seen in the detention percentage. This suggests that there is a slow but active re-suspension and conveyance of sediment occurring within this pond.

When considering NGP pond average results, the sand appears to be more effectively detained in the early events, but the detention efficiency declines severely over extended multiple rainfall-runoff events (month 2 and 3). The clay is detained slightly less efficiently (10% lower than sand) during the initial events (month 1) but the detention efficiency for clay remains functional (80% < efficiency >20%) for a longer period of time/greater number of rainfall-runoff events. The reason for this extended detention efficiency period may result from the preferential detention locations for clay material in this pond, discussed in Section 5.6.2. Overall, the pond is found to provide functional sand and clay sediment detention for a short period of time, the initial 1 to 2 months after release into the pond. Over multiple rainfall-runoff events the sediment is seen to be conveyed through the pond and released into the receiving waterway. Therefore, this pond is found to provide initial temporary sediment detention but limited long term detention. The pond effectively delays and potentially thus dilutes the urban sediment pollution reaching the downstream waterway, but does not effectively prevent sediment transportation into the waterway.

5.6.2 Sediment deposition locale

Sediment samples, from bed deposition and surface flow, were collected across the NGP pond (access prevented sample collection from the J4M8 pond). Samples were taken from the adjacent to the inlet culvert, upstream and downstream of the outlet structure, upstream and downstream from the pond outlet and at 4 locations within the pond. Figure 5.21 illustrates the pond sampling locations and Figure 5.25 presents the sediment deposition results.

Stormwater entering the pond, via the inlet sample pond, generally flows towards sample 3 then moves along the flow path towards sample 5 and down to the outlet. Samples 4 and 5 lie within an area of dense reed vegetation, and this causes some backflow, especially at higher water levels and inflow, towards sample 1. Similarly, sample 4 is located in an area of dense vegetation, equivalent to sample 5, but lying outside the flow path and leading to higher potential deposition in the slackwater environment. Flow over sample 4 occurs primarily during higher inflows and water levels, similar to flow occurrence over sample 1.

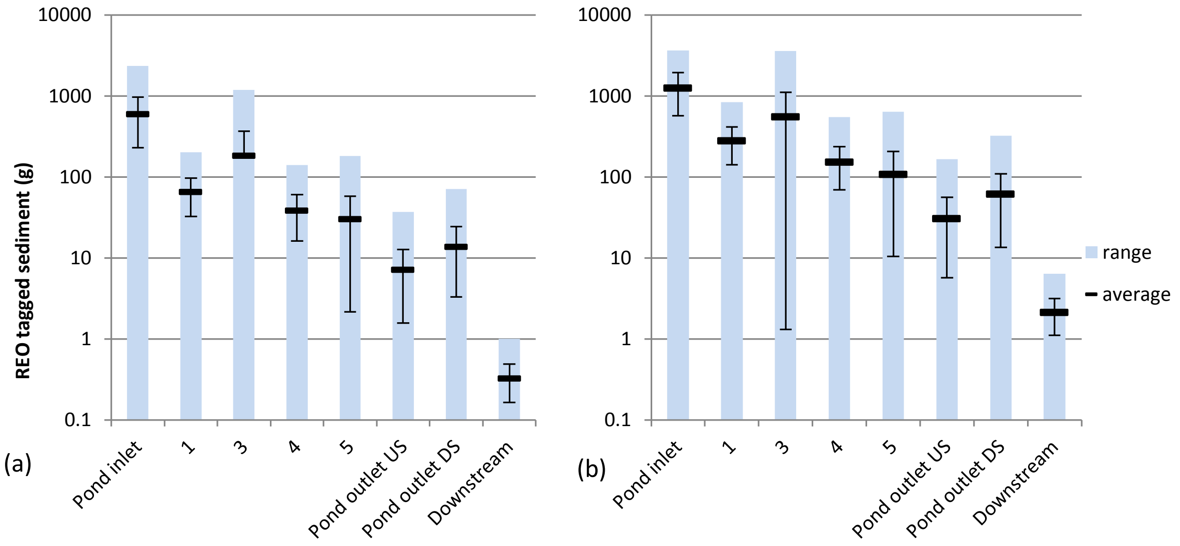


Figure 5.26 Network 4 clay (a) and fine sand (b) preferential deposition locations in the pond (release 1). REO tagged sediment results are representative of sample period deposition across the sample specific SuDS area.

The first observation from the sediment deposition results is that the preferential deposition location for both sand and clay material is close to the stormwater inlet into the pond. Sediment deposition is seen to generally decrease along the flow path through the pond, with a quantity of tagged sediment found in the watercourse downstream (on a downstream receiving waterway sediment bar). The deposition pattern for clay and sand are similar, showing an elevated deposition at sample 3. Deposition at sample 3 is expected, as this area is the main open water section and is surrounded by dense reeds. Flow proceeding through the pond would therefore be conveyed through a vegetated flow path downstream from sample 3, resulting in a vegetation filtering effect and greater deposition rate at sample 3 due to this shift in hydraulic roughness.

There is an interesting trend in deposition around the pond outlet. While both the upstream (US) and downstream (DS) pond outlet results are lower than all within pond locations, the pond outlet DS shows a slightly higher deposition than the US outlet monitoring location. The pond outlet DS is located adjacent to the gabion outflow, but within a setback area of the main channel. Thus the main receiving watercourse flow is conveyed past rather than into this set back area and thus this outflow channel acts similar to a back flow or offset channel section. This may be due to the outlet design, a semi-permeable gabion outlet (rather than a piped outlet) and/or to the dense vegetation at the pond outlet and backflow occurring in this dense vegetation from the downstream receiving watercourse. The use of a gabion weir outlet results in a permeable cross section (rather than a limited discharge orifice such as a pipe outlet) which provides less flow control and a permeable flow path the full depth of the cross section limiting upstream deposition (piped outlets are generally set above bed level). The downstream outlet sample therefore receives flow and sediment from the pond as well as from the receiving waterway (backflow into the outlet area). This results in a low velocity area (meeting of flows moving in opposing directions) and settling of conveyed sediment within the vegetated areas of this outlet zone. The outlet DS results support the sediment detention efficiency results (Figure 5.25), showing that tagged sediment is being conveyed through and out of the pond. The pond outlet DS results are considered a more reliable outlet response than the downstream values due to the limited dilution of sediment by the receiving watercourse at this location.

The preferential deposition locations within this pond are illustrated to be adjacent to the pond inlet and within the open water section of the pond. It is suggested that the boundary effect of the dense reed vegetation surrounding the open water at sample location 3 influences this deposition pattern.

5.6.3 Shift in particle size distribution

The modal particle size for sediment in suspension and deposited at the inlet and outlet of the J4M8 pond have been identified and are presented in Figure 5.27.

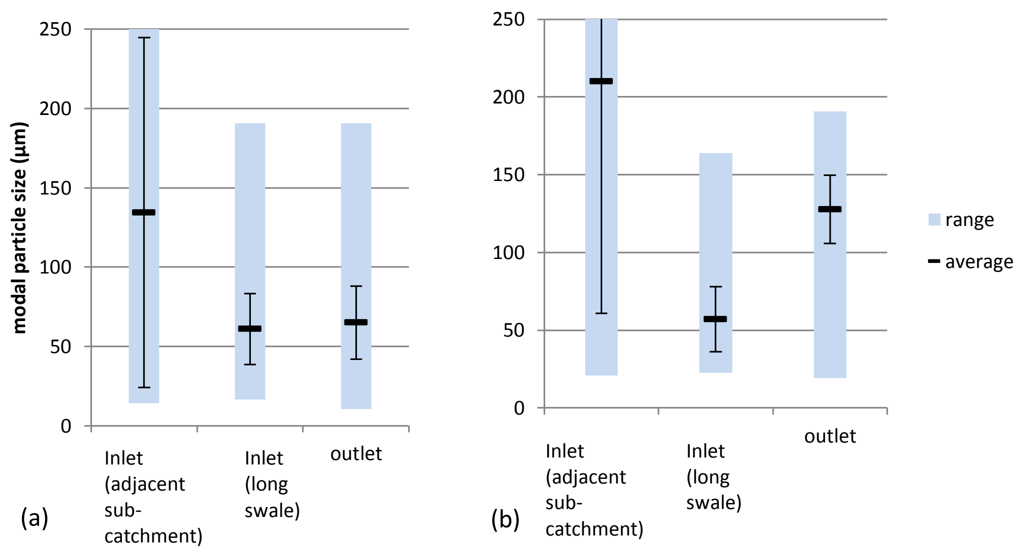


Figure 5.27 Modal particle size for J4M8 pond inlet and outlet. Figure 5.26 illustrates the average suspended (a) and deposited (b) modal particle size for flow and deposited sediment at the pond inlet and downstream from the outlet. Figure 5.27 (a) and (b) inlet range extends to 1375μm and 1921μm.

The J4M8 pond is found to have limited influence on the suspended sediment modal particle size when comparing the long swale inlet to the pond outlet (Figure 5.27a comparison of inlet (long swale) and outlet results). This is because the particle size of suspended sediment at the downstream extent of Networks 1, 2 and 3 (after flowing through multiple SuDS assets) is already small (~ 60µm), and it would be expected that re-suspended sediment discharging from the pond would also be small (smaller sediment is suspended and conveyed more easily, with less energy, than larger material). The modal particle size of adjacent sub-catchment suspended sediment is higher (~135µm); the sub-catchments have limited upstream stormwater management for water quality improvement and collect stormwater from a distribution warehouse (Aldi) and construction site. The pond achieves a decrease in modal particle size (comparison of adjacent sub-catchments to outlet), suggesting the larger suspended particles from the adjacent sub-catchments become deposited within the pond and there may be conveyance of fine sediment and downstream deposition of aggregated cohesive sediments..

The modal particle size of sediment deposited at the inlet and downstream of the outlet of the J4M8 pond shows a change in particle size (Figure 5.27b). Material deposited at the downstream extent of the long swale has a low modal particle size, while material from the adjacent sub-catchments is larger. Sediment deposition downstream from the pond outlet is larger than the long swale but smaller than the adjacent sub-catchment modal particle size. This suggests that overall the pond provides a decrease in deposition modal particle size, detaining larger particles within the pond.

The modal particle size of deposited and suspended sediment across the NGP pond has been similarly defined. Figure 5.28 illustrates that average, standard error and range of modal particle size for each sample location across the pond.

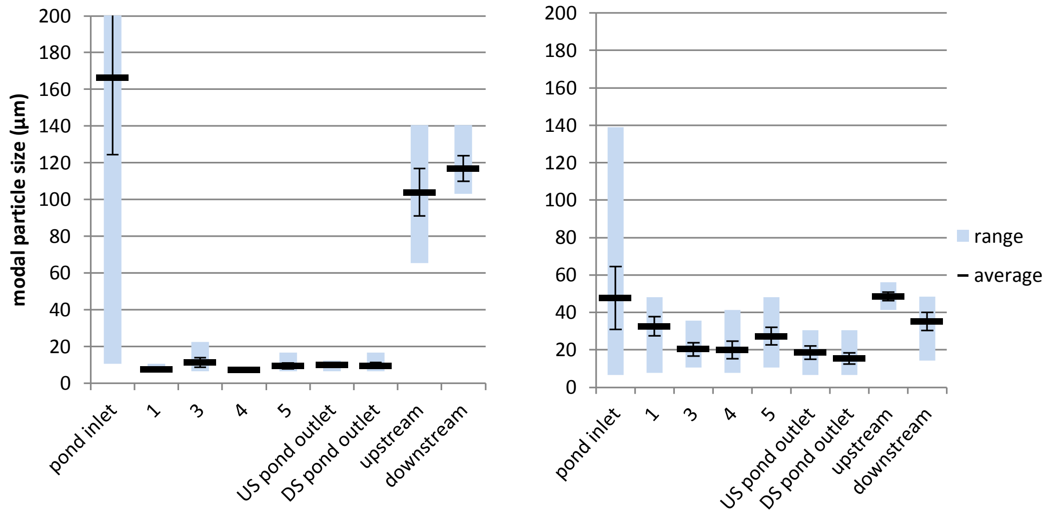


Figure 5.28 Modal particle size for the pond suspended (a) and deposition (b) sediment samples

Larger modal particle sizes are found in suspension and deposited adjacent to the pond inlet (Figure 5.28b). The suspended sediment particle size at the pond inlet is significantly higher than the deposition due to the elevated mixing of sediment by inflow at this location (samples were collected at the edge of the inlet culvert concrete apron) and therefore greater sand in suspension. However, the remainder of the suspended sediment sample particle size results show a smaller suspended particle size compared to bed deposition results. The open water sample, sample 3, illustrates the lowest deposition ratio of clay to sand within the pond (1.52 sand:clay), while all other sample locations within the pond show a significant preference for sand deposition (sand to clay ratios' of >1.8). This suggests that inlet zone and open water are the more effective clay deposition locations in this pond. The higher deposition may be due to collision of fine sediment (clay and silt) within open water areas and thus greater flocculation and floc particulate deposition. The suggested primary methods of clay detention within the pond are therefore flocculation through mixing and boundary effect driven settling rather than vegetation filtering or standard sediment settling processes.

The clay sized sediment appears to be preferentially conveyed through the pond. This is illustrated through the preferential deposition of sand sediment across all monitoring locations (a sand:clay ratio>1 when compared to the release composition). Furthermore, the sediment deposition in the receiving waterway shows an decrease in particle size downstream from the pond outlet, potentially as a direct result of clay released from the pond (confirmed through the REO tag results presented in Figure 5.24 and 5.26). Thus, despite the potential dilution of discharged pond sediment by the receiving waterway, the pond release appears to influence the receiving waterway sediment particle size distribution (showing greater fine material upstream of the pond).

5.6.4 Pond conclusions

A summary of the pond sediment conveyance and detention efficiency research findings is provided. The key new science reflecting wetland fine sediment transport discussed in this section of Chapter 5 are summarised below.

Sediment is transported through both the NGP and J4M8 pond over multiple rainfall-runoff events. Tagged find sediment is found within the receiving waterway downstream of the NGP pond and downstream of J4M8 pond outlet. Thus both ponds provide temporary detention with delayed sediment discharge to the downstream receiving waterway.

The sediment detention efficiency of the NGP pond is initially functional (≥ 20% but <80%) but declines significantly over multiple events (month 2 onwards) to below 20%.

The J4M8 pond show high efficiency (all results >80%) but indicates a slow decline over the monitoring period (3% decrease in detention efficiency over 12 months of monitoring).

The preferential deposition locations within the NGP pond are at the inlet and within the open water section of the pond. The pond does not primarily act as a vegetation filter (a semi-porous asset – illustrated in the linear wetland results) but relies of flocculation through mixing and boundary effect sediment settling to achieve sediment detention. Clay is preferentially deposited adjacent to the NGP pond inlet, while a greater proportion of sand is deposited at all other monitored locations across the pond.

The J4M8 pond decreases the suspended and deposited sediment modal particle size (in general). The decrease in modal particle size illustrates the pond detention function, removing larger sediment from the stormwater flow. Sediment particle size decreases through the NGP pond, illustrating an increasing detention of finer sediment along the pond flow path.

The NGP pond receiving waterway sediment shows a decrease in deposited particle size but an increase in suspended sediment particle size downstream from the pond. The pond discharge therefore influences the waterway sediment particle size distribution.

5.7 Asset comparisons

Using the tagged sediment methodology developed specific to this thesis (Chapter 3), long-term sediment tracing and detention efficiencies for four (4) SuDS asset types (wetland, linear wetland, swale, pond) has been realized and proven, over 6-12 months, to demonstrate significant ongoing downstream conveyance of urban sediment to discharge into natural watercourses .

5.7.1 Sediment transport and deposition trends

From Sections 5.1-5.6 shows tagged sediment concentration in suspended and bed deposition samples to fluctuate over the 12 month monitoring period, illustrating ongoing (re)suspension, deposition, entrainment and transport. All asset detention efficiencies decreased over the monitored timeframe, demonstrating the hierarchy of detention efficiency NGP pond (<4%) < wetland (46%) < swale (69%) < linear wetland (70%) < J4M8 pond (95%) from the statistics provided in Table 5.5.

Table 5.5 SuDS asset sediment detention efficiency (%) comparative overview. The literature review expected detention from design guidance (^) efficiencies are taken from Tables 2.3-2.6 in Chapter 2.

Monitoring period	release	month 1	month 2	month 3	month 4	month 5	month 12	Literature review reported or expected detention^
Wetland – average								
average	0	94%	90%	88%	78%	75%	46%	75-92%
Linear wetland - average								
average	0	79%	75%	72%	78%	73%	70%	70-92%
Swale - average								
average	0	84%	82%	82%	84%	82%	69%	76-84%
Pond – total tagged sediment release								
NGP average	0	75%	41%	12%	7%	4%	-	60-90%
J4M8 average	0	98%	97%	97%	96%	96%	95%	

Note: the average values presented in this table are taken from the SuDS asset overarching average detention efficiencies presented in Tables 5.1-5.4 of Chapter 5. The total range of detention efficiencies for each of these assets is presented in their respective tables (Tables 5.1-5.4).

This suggests that the NGP may be a temporary deposition asset of urban sediment, but may function as a sediment source (conveying sediment) to downstream waterways. The wetland, linear wetland and swales illustrate greater efficiency in sediment detention, arguable acting as both a sink (detaining) and source (conveying sediment to downstream SuDS assets or possible waterways). Thus, the REO analysis results presented in Table 5.5 could tentatively illustrate the monitored SuDS assets efficiency in source and sink function (sliding scale from 100% detention=sink to 100% conveyance=source).

From Table 5.5 it is very evident that, when compared to existing literature of 'expected' or reported SuDS efficiencies, only the large J4M8 pond and site's linear wetlands are able to maintain this level of functionality over the first year of rainfall-runoff events (after initial sediment release). That said, the linear wetland detention efficiency falls to the minimum (70%, Table 5.5) 'expected' performance value of the literature after 1 year; thus, over an extended period, such as the operational life of a system (25-30 years) further decline of sediment detention is expected from extrapolation of the temporal trend observed in Table 5.5 (Section 5.8.4). Conversely, the perennial wetlands and small SuDS ponds (NGP) prove effective (to the literature minima, Table 5.5) for only very short term sediment detention of 1-5months. Hence, small perennially wet SuDS assets appear to underperform most, compared to ephemeral and very large perennial assets.

5.7.2 Preferential deposition locations

The preferential sediment deposition within each of the monitored SuDS assets does not present a consistent trend. Sediment deposition appears to be influenced by asset design, specifically the provision of open water and dense vegetation. Downstream boundary conditions also appear to influence sediment deposition within SuDS. Overall, there are three aspects of asset design that appear highly influential of preferential deposit location (Sections 4.3.2 (total mass deposition), 5.3-5.6 (REO tagged sediment deposition)): (i) dense emergent vegetation, especially promoting upstream deposition in linear wetlands and at pond margins; (ii) decelerating flow in deep open water, for natural settling at wetlands and pond inlets; (iii) ephemeral flow path vegetation density (hydraulic roughness), promoting downstream sediment deposition in the swales; and (iv) outlet design (flow discharge into a downstream wet SuDS asset or an ephemeral/dry asset/flow path) resulting in downstream flow velocity decrease and therefore sediment deposition (as illustrated in the long swale (Figure 5.19).

5.7.3 Particle size distribution influence

SuDS asset have been illustrated to modify the particle size distribution and modal particle size of suspended and deposited sediment. All assets indicate preferential detention of finer particles, thus a decline in the modal particle size of deposited sediment moving downstream through the SuDS asset. Departure from this general trend appear specific to inlet/outlet asset design effect on local fluid dynamics, as seen in the wetland where surface runoff into the wetland results in extended fine sediment suspension within the wetland flow path.

Suspended sediment particle size distribution generally presents an inverse trend to bed deposition data. The exception is the NGP pond; where there is a decline in sediment modal particle size found for both bed deposition and suspended sediment. This may be due to the a continuous flow and higher rate of sediment conveyance through this pond, resulting in preferential detention of larger sediment particles but not to such an extent that this results in and an increase in suspended sediment modal particle size (a loss of fine sediment from suspension).

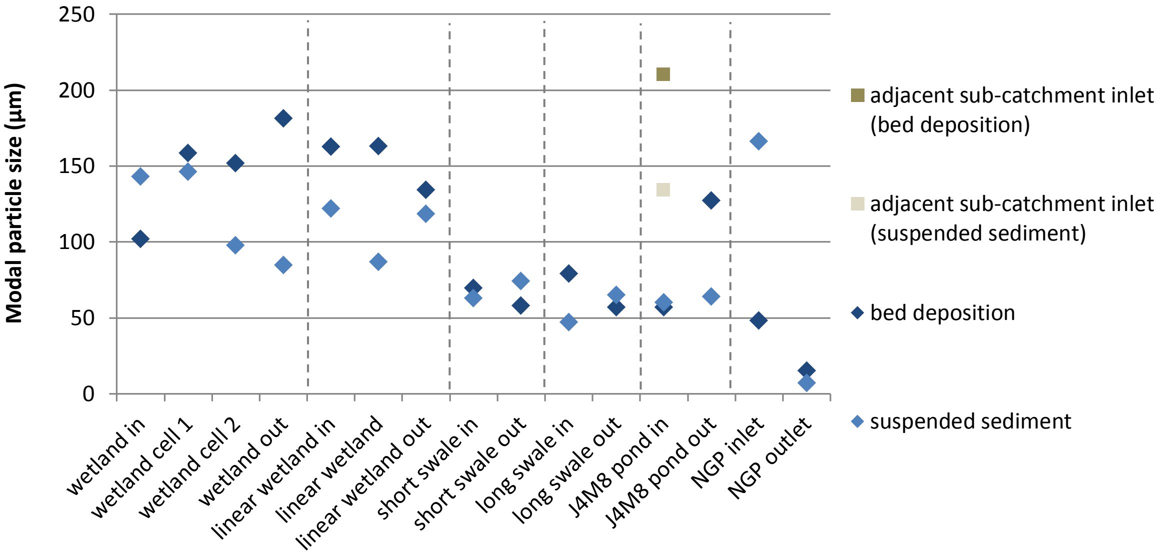


Figure 5.29 Comparative overview of sediment sample modal particle size (all monitored assets)

Overall, the monitored SuDS assets illustrate a decrease in modal particle size (Figure 5.29). The influence, a graphical displacement towards greater/lower clay sized material in the discharge from each asset, was not consistent across all SuDS asset types, but is strongly influenced by asset design, namely provision of open water and dense vegetation within the flow path. The occurrence of cohesive aggregation in these locations may be influential in deposition of silt and clay sediment, supporting the decrease in modal particle size within SuDS assets and along SuDS networks (treatment trains).

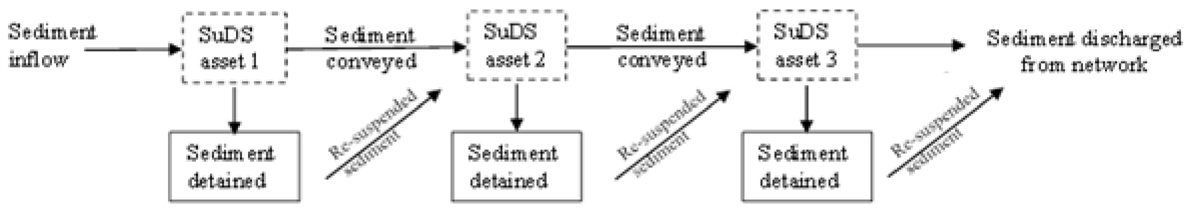
5.8 Network SuDS sediment detention efficiency

5.8.1 Network sediment detention efficiency over the monitoring period

REO sediment tracing has illustrated the multiple event sediment transport and detention efficiency of individual SuDS assets (Section 5.1-5.6). However, SuDS assets are often placed in a treatment 'train', i.e. a linear series of connected assets, to provide cumulative hydraulic and sediment pollution benefit(s) (Woods-Ballard et al. 2015, SPP 2014).

To assess whether the SuDS network is greater than the sum of its parts, the individual SuDS sediment detention efficiency results discussed in Sections 5.1-5.6 have been compared to the overall network results. To calculate the cumulative sediment detention as a percentage detention from each asset overall sediment release, the following general formula was used Equation 12 (and Figure 5.30).

Figure 5.30 Schematic of cumulative network sediment detention calculation



$$S_{out_{t_i}} = S_{in_{t_0 \to t_i}} - \sum_{t_i}^{SuDS} S_d, S_s, -S_r \qquad \text{Eqn.12}$$

Where:
S_{in} = sediment inflow into the SuDS asset/network
S_{out} = sediment discharged from the SuDS asset/network
t_0 = time of sediment release of initial inflow of sediment into the SuDS asset/network
t_i = time of sample (point in time for which Y, the sediment discharge, is being calculated
S_d = sediment detained by the individual SuDS asset (e.g. wetland, linear wetland, swale)
S_r = re-suspended bed deposition the individual SuDS asset
S_s = sediment in suspension within the individual SuDS asset
SuDS = SuDS asset of interest

Therefore, in a network comprised of two SuDS assets where SuDS asset 1 had a sediment detention efficiency of 80% (0.8), then the sediment discharged from asset 1 is 1 - 0.8 = 0.2 i.e. 20% of the total inflow load is conveyed to asset 2. Following this, if asset 2 is 50% efficient, half (0.5) of the inflow to this asset is detained and half (0.5) discharged, such that it discharges (0.2 x 0.5 = 0.1) 10% of the total inflow to the network. As REO data (Section 5.3-5.6) has illustrated that each asset has a different efficiency, then treatment trains of different combinations, sequences and numbers of assets will exhibit different overall network performance. The present section of this thesis considers both, whether the network performs according to the theoretical assumption of Equation 12 and, whether long term network efficiency complies with the literature evidence and best practice design standards stated in Section 2.6).

Thus, this section compares the following:

Asset-based field data used within Equation 12 to provide the calculated network efficiency. For the purpose of this analysis, the downstream extent is the long swale outlet (Section 3.3, Network truncated at the long swale due to limitations of data collection across the J4M8 pond). This is calculated in two ways:

(a) Using the actual measured asset detention data for that particular network; this is defined as "Calculated: Asset N1" (extracted from Tables 5.1, 5.2, 5.3 Network and asset specific average values);

(b) Using the "average" efficiency of all equivalent asset type, independent of network; this is defined as "Calculated: Asset average" (extracted from Tables 5.1, 5.2, 5.3 asset average values).

The four sediment release and monitoring datasets created for each SuDS asset provide four network sediment detention efficiencies (for 3 months (Release 4) to 12 months (Release 1)). A trend analysis (line of best fit) was created to estimate the potential 12 month Network sediment detention efficiency if all four REO dataset were considered (rather than solely Network 1). These trendlines are presented in Figures 5.31 and estimated 12 month sediment detention efficiencies presented in Table 5.6 as "Calculated: Network trend".

The literature-based evidence and design guidance standards for network efficiency.

Employing network analysis methods (1) to (3), the total network sediment detention efficiency, 12 months after initial sediment release onto the supply (urban) surface, have been calculated. The network analysis has been limited to Networks 1, 2 and 3 as they incorporate multiple SuDS assets (Network 4 provides a single SuDS asset only, Section 3.2). The calculated network sediment detention efficiencies are presented in Table 5.6 and Figure 5.31 (a-c).

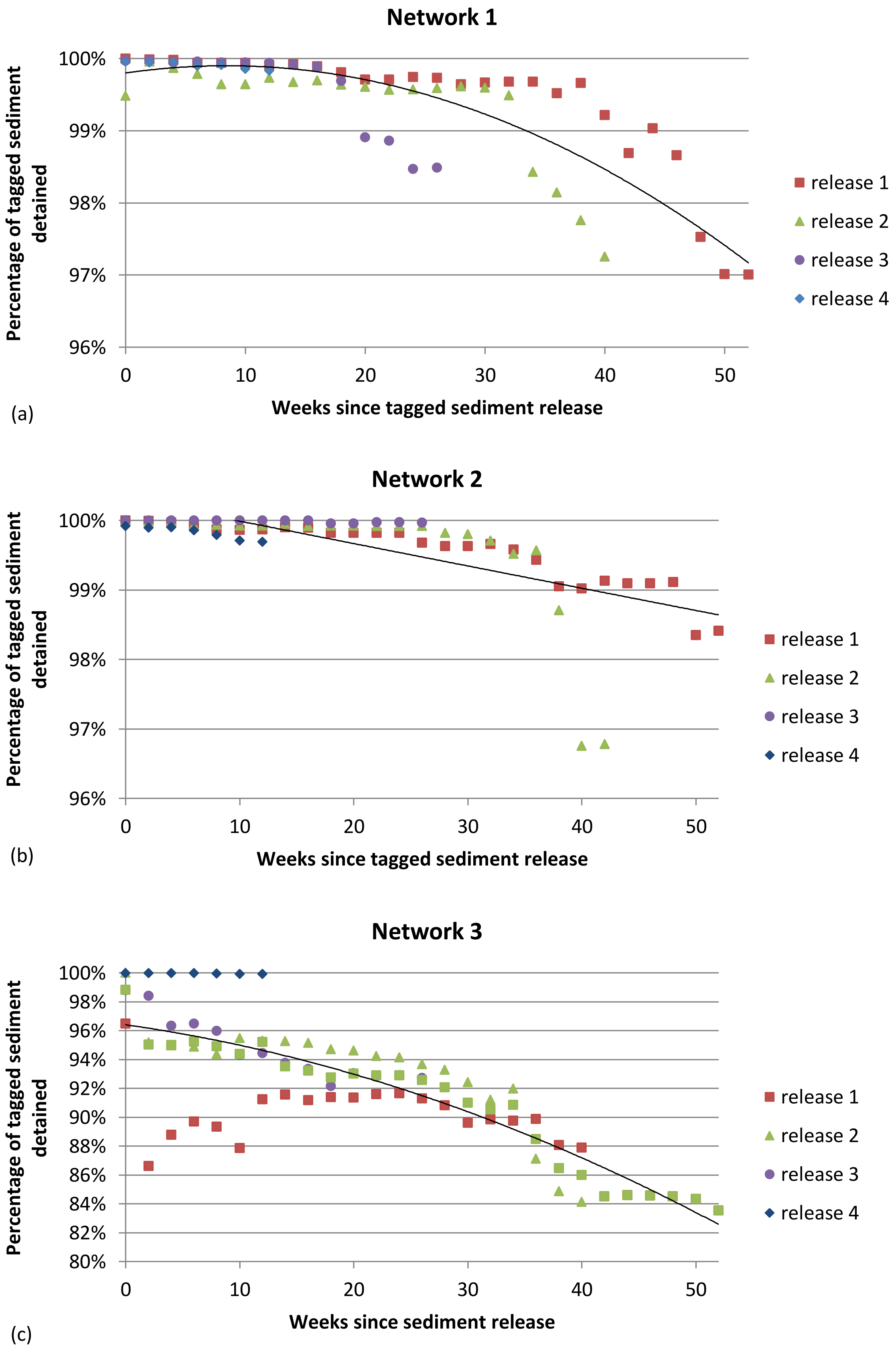


Figure 5.31 Multiple repetition sediment detention efficiencies for Network 3. The average trend is illustrated as a black line, while Release 1-4 are shown as colour coded points.

Table 5.6 Summary of Network sediment detention efficiencies. The four methods of network sediment detention calculation are described in Section 5.8 (1) to (3).

Network		SuDS assets making up the Network			Cumulative network calculation
		Wetland	Linear wetland	Swale	
Network 1	Calculated: Asset R1	19%	75%	84%	97%
	Calculated: Asset average	46%	70%	69%	95%
	Calculated: Network trend				98%
	Literature guidance[3]	90%	90%	76%	99%
		Wetland	Linear wetland	Swale	
Network 2	Calculated: Asset R1	73%	64%	79%	98%
	Calculated: Asset average	46%	70%	69%	95%
	Calculated: Network trend				97%
	Literature guidance	90%	90%	76%	99%
		Short swale	Long swale		
Network 3	Calculated: Asset R1	71%	45%		84%
	Calculated: Asset average	71%	69%		91%
	Calculated: Network trend				85%
	Literature guidance	76%	76%		98%

General findings from Table 5.6 comparison of network sediment detention efficiency calculations are presented below:

Network sediment detention efficiency estimated from literature guidance is consistently higher than results calculated from field data. This illustrates that there is value in field validated sediment detention efficiencies as the final network results differ from Guideline expected or reported values even after only 12 months of SuDS operation.

Network sediment detention efficiency calculated from Release 1 results fall between 1-14% lower than Guidance expected or reported values. This illustrates the influence (a decrease in sediment detention efficiency) of multiple rainfall-runoff events over a 12 month period on asset specific and thus network sediment detention efficiency.

Network sediment detention efficiency calculated using asset average results (the average sediment detention efficiency resultant from review of all 4 replicated REO tagged sediment dataset sets for each SuDS asset) illustrate lower network results for Networks 1 and 2, but elevated results for Network 3. This shows the variability from the repeated tagged sediment detention results influence (±5%) in network sediment detention results, demonstrating the need for replication in tagged sediment experiments

[3] Minimum UK guideline expected or reported removal efficiency rates adopted to support conservative estimation

5.8.2 Extended network sediment detention trend extrapolation

Each of the three monitored networks presented in Section 5.8 illustrate a slow declining total sediment detention trend (Figures 5.31 (a-c)). SuDS are constructed with an expected design life of 25-30 years (Woods-Ballard 2015, Water by Design 2006) and it is therefore important to consider sediment detention over this extended period if detention efficiency and sediment deposition levels in the network are to be better informed. To consider the possible 30 year detention for the individual tagged sediment release, the trends found for the three networks can be extrapolated. This assumes future rainfall-runoff, asset characteristics and conveyance processes remain equivalent to the year monitored herein; as such the conclusions drawn from the extrapolation are strongly caveated in this regard. However, this approach does provide a first-pass indicative long term SuDS network detention efficiency, underpinned by medium-term monitored data.

Due to the uncertainty created by extrapolation analysis, four common lines of best fit have been examined (linear, exponential, logarithmic and power laws). From Table 5.7 the linear and exponential equations have the higher R^2 values (~0.5) and are preferred in estimating futuristic detention. In summary of Table 5.7, Network 3 is predicted to operate above the 80% efficiency design standard (Tables 2.4 – 2.6) for only a 1.3 year design life; whilst Networks 1 & 2 illustrate longer term performance with 9-16 year design lives. Overall, the extrapolation results show that all Networks would be functioning at only 0-65% detention efficiency at the end of their 30 year design life.

Table 5.7 Network trends and extrapolated detention results. Rows highlighted “orange” trends with higher R^2 value (R^2 < 0.4) fit and preferentially used for analysis.

	trend form	equation	R^2	80% year	% detention at 30 years
Network 1	linear	$y = -0.0004x + 1.0$	0.57	9	46%
	exponential	$y = 1.0023e^{-4E-0.4x}$	0.57	10	52%
	logarithmic	$y = -0.002\ln(x) + 1.0003$	0.21		
	power	$y = 1.0003x^{-0.002}$	0.21		
Network 2	linear	$y = -0.0003x + 1.0$	0.49	15	58%
	exponential	$y = 1.002e^{-3E-0.4x}$	0.48	16	65%
	logarithmic	$y = -0.001\ln(x) + 1.0006$	0.17		
	power	$y = 1.0006x-0.001$	0.17		
Network 3	linear	$y = -0.0025x + 0.9732$	0.60	1.3	0%
	exponential	$y = 0.9739e^{-0.003x}$	0.60	1.3	1.5%
	logarithmic	$y = -0.019\ln(x) + 0.9718$	0.38		
	power	$y = 0.9716x^{-0.021}$	0.37		

Further longer term monitoring needs to be undertaken to examine the accuracy of these extrapolations, however the field monitoring results suggest that SuDS stormwater management measures should be considered as temporary detention measures rather than sediment sinks.

5.8.3 Key network detention efficiency analysis conclusions

The key network analysis findings have been summaries below:

- The network sediment detention trends show a slow decline in detention efficiency over multiple events;
- Extrapolation of network sediment detention trends suggests a longer term efficiency of less than guideline expected or reported efficiencies (decline after 30 years of <65%); and,
- The cumulative network sediment detention efficiency is more effective than any one of the individual SuDS components within the network. The network can be considered greater than the sum of its parts.

5.9 Chapter conclusions

The key finding from the REO tracer field experiments presented within Chapter 5 is that sediment is not permanently retained within SuDS assets but continues to move once initially introduced to the SuDS asset. Furthermore, the rate of detention efficiency within SuDS assets is not constant, but declines over multiple rainfall-runoff events. The rate of sediment detention efficiency decline appears to vary between SuDS asset but all asset illustrate a decrease in efficiency over the extended monitoring period. A summary of the key findings is provided below:

- The sediment detention efficiency fluctuates over the entire monitoring period, showing a decline overall.
- Compared to the guidance expected or reported SuDS efficiencies, the monitored SuDS assets generally illustrate 'as expected' efficiencies for the early rainfall-runoff events (i.e. for the initial event and first month of results, with exception of the NGP pond), but over the extended monitoring period (12months for networks 1-3, 6 months for network 4) the guidance expected or reported efficiencies overestimated actual SuDS performance
- SuDS assets are have been shown to modify modal particle size. J4M8 and NGP ponds, linear wetland decreases the modal particle size of deposited and

suspended material, swales decrease the deposited but increase the suspended modal particle size, and the wetland is found to decrease the suspended but increase the deposition sediment modal particle size.

- Over the total network (Networks 1-4) the modal particle size is found to decrease, indicating detention of larger sediment (sand) over smaller particles (silt/clay).
- Sediment detention efficiency for networks (stormwater flow paths comprised of more than one SuDS asset) are found to be greater than for individual assets.
- Tentative extrapolation of network sediment detention efficiency trends suggests that longer term (life cycle) efficiencies may be notably lower than published guidance expected or reported efficiencies (<80%).